Social-ecology of rural abandonment: from farmers' perceptions to ecosystem services

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Other papers and chapters

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Dedication

To my son, Matias. Thank you pimpolho.

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Abstract

Along the last fifty years, we have witnessed a gradual conversion of agro-ecosystems into abandoned land throughout much of Europe. This has raised many concerns on how to manage areas that have undergone abandonment as well as those areas that may be facing future land abandonment. This phenomenon has been attributed to a complex set of ecological, social, economic and political drivers acting in spatially and temporally diverse patterns. In this context, the research developed for this thesis aimed to provide information for supporting decision and policy makers as well as for developing land management strategies in those areas undergoing land abandonment.

We first identified those pressures and frictions responsible for inducing and halting agro-pastoral abandonment, and determined local residents’ willingness to leave a rural mountain community in northern Portugal. The analysis revealed that both pressures and frictions varied along a temporal gradient. Our findings also unveiled that certain pressures and frictions have a tipping point at which time a pressure becomes a friction and vice-versa. Local willingness to leave was found to be directly correlated to individual’s farmland utility in the rural community. In other words, those farmers rearing large numbers of livestock are less willing to leave than those with low numbers of livestock. We also determined an inter-relationship between drivers. For example, farmers receiving high subsidies usually carry out a more modern type of agriculture.

In a second study, we identified the existence of differences in the human preference of ecosystems and their services along an urban-rural gradient in another mountain area in Portugal. In addition, we identified local’s perception on landscape change and on possible conservation strategies. Significant differences were found for several perceptions along the rurality gradient. Our results also revealed that residents relate conservation value to aesthetic appreciation. These results provide landscape planers with a better understanding of how local stakeholders’ preference for ecosystems and their services vary spatially and how those perceptions are reflected on the landscape.

In a third study we aimed to determine how land abandonment may have impacted ecosystem services such as carbon storage/sequestration, soil conservation, water yield, and biodiversity. We followed a backcasting approach using a time series of land cover maps reflecting a trend of land abandonment in a mountain rural landscape. We found an increasing trend for all focal ecosystem services, except for biodiversity which suffered a decrease in plant taxa with a higher affinity for agricultural systems.

Finally, in a fourth study we identified, at the European scale, the spatial distribution of ecosystem services and the benefits of rewilding those areas that may be facing future land abandonment, using high quality wilderness as a proxy for rewilding. Mountain systems were identified as high suppliers of ecosystem services and the promotion of land abandoment (rewilding) was predicted to enhance the capacity of ecosystems to supply regulating and cultural services, such as carbon sequestration and recreation.

The ensemble of results obtained through this research provides valuable clues for future research priorities as well as for improved policy frameworks for adaptive management of land abandonment processes, thereby supporting and enhancing the supply of multiple ecosystem services.

Keywords: drivers of land use, ecosystem services, human preference, InVEST, land abandonment, migration, rewilding, socio-economic dynamics, urban-rural gradient

Resumo

Ao longo dos últimos cinquenta anos, temos assistido a uma conversão gradual de ecossistemas agrícolas em terras abandonadas em grande parte da Europa. Este fenómeno tem levantado muitas preocupações sobre como gerir as áreas que sofreram abandono assim como as áreas que poderão enfrentar abandono no futuro. Este fenómeno tem sido atribuído a um conjunto complexo de factores ecológicos, sociais, económicos e políticos, que atuam de forma espacial e temporalmente diversa. Neste contexto, a investigação desenvolvida para esta tese teve por objetivo fornecer informações para suporte à decisão técnica e política, bem como para o desenvolvimento de estratégias de gestão para as áreas submetidas a abandono.

No primeiro estudo, procurámos identificar as pressões e os atritos responsáveis pela promoção ou prevenção do abandono agro-pastoril, e ainda determinar a disposição dos moradores para abandonar uma comunidade rural de montanha no norte de Portugal. A análise revelou que as pressões e os atritos variaram ao longo de um gradiente temporal. Os nossos resultados também revelaram que algumas pressões e atritos têm um ponto de inflexão em que a pressão se torna um atrito, ou vice-versa. A disposição dos residentes para abandonar a comunidade está diretamente relacionada com a utilização individual da terra. Assim, os agricultores que criam grande número de animais estão menos dispostos a abandonar do que aqueles com baixos níveis de gado. Também determinámos inter-relações entre os diversos factores. Por exemplo, os agricultores que recebem mais subsídios tipicamente praticam uma agricultura mais moderna.

Num segundo estudo, identificámos a existência de diferenças na preferência humanas dos ecossistemas e dos seus serviços, ao longo de um gradiente urbano-rural numa outra área de montanha em Portugal. Além disso, avaliámos a percepção humana das alterações na paisagem bem como sobre possíveis estratégias de conservação. Foram encontradas diferenças significativas para várias percepções ao longo do gradiente de ruralidade. Os nossos resultados também revelaram que os moradores relacionam o valor de conservação com a apreciação estética. Estes resultados fornecem informação para uma adequada estratégia de conservação da paisagem, baseada numa melhor compreensão dos atores locais, nomeadamente como variam espacialmente as preferências dos ecossistemas e dos seus serviços, e como as suas percepções se refletem na paisagem.

Num terceiro estudo procurámos determinar como o abandono da terra terá vindo a afectar serviços de ecossistema como o armazenamento/sequestro de carbono, a conservação do solo, o fornecimento de água e a biodiversidade. Adoptámos uma abordagem “backcasting” com base numa série temporal de mapas de ocupação do solo que evidenciava uma tendência para o abandono numa paisagem rural de montanha. Encontramos uma tendência crescente na provisão de todos os serviços de ecossistema avaliados, com exceção para a biodiversidade, a qual sofreu uma diminuição na diversidade de plantas com maior afinidade para as áreas agrícolas.

Finalmente, num quarto estudo identificámos, à escala europeia, a distribuição espacial de vários serviços de ecossistemas, assim como os benefícios gerados pela renaturalização das áreas que poderão enfrentar abandono no futuro. Foi utilizado o indicador “high quality wilderness” como “proxy” da renaturalização. Os sistemas montanhosos foram identificados como importantes fornecedores de serviços de ecossistema, e a promoção da renaturalização poderá vir a aumentar a capacidade de os ecossistemas fornecerem serviços de regulação e culturais, como o sequestro de carbono e a recreação.

O conjunto de resultados obtidos nesta investigação fornece pistas valiosas para a definição de prioridades de investigação futura e para a definição de melhores políticas para a gestão adaptativa do abandono rural, promovendo dessa forma a provisão de múltiplos serviços de ecossistema.

Palavras-chaves: abandono rural, determinantes do uso da terra, dinâmicas socioeconómicas, gradiente urbano-rural, InVEST, migração, preferência humana, renaturalização, serviços de ecossistema

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Chapter 1. Introduction

1.1. Biodiversity, livelihoods and traditional ecological knowledge in organic farming systems

1.1.1. Traditional agricultural landscapes

Local communities in rural mountain landscapes have lived off the land for centuries, developing, using and improving Traditional Ecological Knowledge (TEK) as the primary approach to manage natural resources while sustaining biodiversity and controlling the overall ecology of landscapes (Gadbil and Berkes 1991). Traditional Ecological Knowledge has been defined as the knowledge acquired through extensive observation and experiences, passed down through generations. Along the years there has been a growing interest in TEK, primarily in its contribution to the conservation of biodiversity (Gadgil et al. 1993, Berkes et al. 2000). Studies have shown that this ancestral knowledge has permitted the persistence of social-ecological systems worldwide as well as their adaptation to novel environmental and climatic challenges (Garray and Larrabure 2011). In the circumpolar north, wilderness protection and restoration are essential in maintaining traditional relationships with nature (Watson et al. 2013), whereas studies in Mediterranean rural mountain areas have underlined the ecological function agricultural terraces have played in soil conservation (Gallart et al. 1994; Lasanta et al. 2001).

Nonetheless, empirical studies have shown that the perseverance of traditional knowledge has been tested throughout much of Europe as rural mountain landscapes have undergone dramatic social and demographic changes leading to discontinuation of traditional practices (MacDonald et al. 2000; Baldock et al.1996; Keenleyside and Tucker 2010). These changes have complicated the transmission of traditional knowledge from one generation to the next. Currently, the rate of erosion of traditional knowledge in rural communities is largely unknown, as are its consequences to environmental conservation.

Historically, we can identify five stages of evolution of European landscapes: the natural, prehistoric landscape; the antique landscape; the medieval landscapes; the traditional agricultural landscapes; and the industrial landscapes (Plieninger et al. 2006). Traditional agricultural landscapes date back to the Renaissance and in some cases have reached present times (Plieninger et al. 2006). Anthropogenic disturbances played a fundamental role in shaping landscapes (Anthrop 2005), and were/are the primary inducers of spatial landscape patterns. Traditional landscapes in Europe are mainly associated to agricultural uses, shaping complex and biodiverse systems, many times characterized by their physical constrains (Plieninger et al. 2006; McDonald et al. 2000), such as altitude, slopes, soils, and climate.

Traditional land uses have been defined as practices which do not use modern technology (Bignal and McCracken 1996). They are normally characterized as low intensity systems, which include livestock, arable and permanent crop and mixed systems (Baldock et al. 1993; Bignal and McCracken 1996). In some cases these land uses are labor intensive, but they include low nutrient input and production is mainly for subsistence. These systems vary greatly across Europe. Mountainous regions of northern Portugal typically present an agro-pastoral system (Calvo-Iglesias et al. 2009; Mottet et al. 2006) which combines the practice of growing crops and the raising of livestock, where semi-natural meadows and common land are used for hay and grazing, and cropland is located closer to the villages (Figure 1.1).


Figure 1.1 Traditional agricultural practices in a northern rural mountain community in Portugal-Castro Laboreiro (Photos: Yvonne Cerqueira).

As described by Baldock et al. (1993), low-intensity farming systems are characterized by low input of chemicals and by management practices in which farming allows or contributes to sustaining species and habitats of European conservation value, many of which are protected under the European Union Habitat Directives (Directive 92/43 EEC) (Andersen 2003). They are often associated to the High Nature Value (HNV) farmland concept (Paracchini et al. 2008), which includes three different landscape types: 1) farmland with a high proportion of natural and semi-natural vegetation, 2) farmland with a fine mosaic of habitats and/ or land uses, and 3) farmland supporting rare species or a high proportion of European or world populations (Andersen et al. 2003). The origin and maintenance of these HNV farmland, such as grazed uplands, alpine meadows and pastures, steppic areas in eastern and southern Europe, and agro-forestry areas in Spain (*dehesa*) and Portugal (*montado*), require some type and frequency of human intervention. According to Ostermann (1998), 28 of the 198 habitats listed under the European Habitats Directive are threatened by the abandonment of low-intensity agricultural practices. This list has since then been updated to a total of 63 habitat types that benefit from agricultural activities related to grazing and mowing (Halada et al. 2011). The monitoring of trends in HNVF areas is thus a priority as well as a challenge due to the complexity and diversity of rural landscape systems (Lomba et al. 2014).

The World Heritage Convention of 1992 was the first international legal instrument to recognize and protect cultural landscapes (http://www.whc.unesco.org). Presently, 82 cultural landscapes have been inscribed in the World Heritage List. The United Nations Educational, Scientific and Cultural Organization (UNESCO) defines these cultural landscapes as “organically evolved landscapes resulting from an initial social, economic, administrative, or religious imperative”, falling under three main categories. These categories are: (1) defined landscape designated and created intentionally by man; (2) organically evolved landscapes with two sub-categories: (a) a remnant landscape in which an evolutionary process came to an end at some time in the past, but whose significant distinguishing features are still visible in material form, or (b) a landscape that continues to exist in the present day and retains an active social role in contemporary society, closely associated with the traditional way of life; and (3) associative cultural landscapes (Fowler 2003).

Along the years the recognition of the importance of these traditional agricultural landscapes in promoting or enhancing agri-biodiversity has resulted in the implementation of several strategies, policies and conventions at the European level (Paracchini et al. 2008). These include e.g. the Rural Development Policy (Community Strategic Guidelines for Rural Development, Programming Period 2007-2013), the Pan-European Biodiversity and Landscape Strategy (PEBLDS), the Bern Convention, the European Landscape Convention, and the Habitats and Birds Directives. Specifically, the Community Strategic Guidelines for Rural Development (European Council 2006) states that “biodiversity and the preservation and development of high nature value farming and forestry systems and traditional agricultural landscapes” is a EU level priority area to protect and enhance natural resources in rural landscape areas.

1.1.2. The connection between farming, biodiversity and conservation

In the last few decades, the polarization of an extensive agriculture towards intensification or abandonment of traditional activities has threatened biodiversity of farmland habitats, particularly of semi-natural habitats (Baldock et al. 1996). In Europe, grasslands have declined by 12.8% between 1990 and 2003 (FAO 2006). They have been identified as one of Europe´s most species-rich plant communities (Walllis De Vries et al. 2002) providing also habitats to numerous bird and wild fauna populations (Laiolo et al. 2004). Reports revealed that butterfly species have suffered a 28% decline between 1980 and 2002 (EEA 2005). This dramatic decline has been attributed to decreases in grassland habitat types. In Romania, promoting the conservation of traditional silvo-pastoral practices has been considered a priority in the survival of several woodpecker species (Dorresteijn et al. 2013).

Changes in management regimes (i.e. increase/decrease of grazing livestock) have also resulted in a loss of functional groups and ecosystem function (Peco et al. 2012; Klimek et al. 2007). Grasslands, for example, provide a wide range of ecosystem services important for human well-being, such as pollination services, carbon storage, flood prevention, water regulation, biodiversity refuge, educational/scientific and recreational uses. Although there is an increasing recognition of the importance of these and other habitats, for approximately 50% of the Mediterranean agricultural habitats their conservation status is unknown. In the NATURA 2000 network, 76% of grasslands are in unfavourable, inadequate or bad status and for 17% that status is unknown (EEA 2010).

Presently, under Article 17th of the Habitats Directive, all member states are required to survey important habitats and species of conservation status and are required to monitor and report all actions to maintain and restore both habitats and species. EU policies and targets have also, until recently been centered around the “preservation” and maintenance of traditional practices (Baldock et al. 1993; Baldock et al. 1996, Paracchini et al. 2008; Kleijn and Sutherland 2003; Halada et al. 2011; Fischer et al. 2012). Land use changes and mountain agriculture are targeted by different measures in the Common Agricultural Policy (CAP) such as agri-environmental schemes. The creation of financial incentives, such as,agri-environmental payments, has been the primary strategy applied to motivate and promote the continuation of traditional practices while benefiting biodiversity. These incentives mainly target LFA (Less Favored Areas) providing extra economic aid to areas which due to natural constraints are unable to intensify their production. However, one of the main challenges in preserving priority habitats is the lack of information on location and condition of these ecosystems as well as the reoccurring issue on defining permanent pastures and semi-natural grasslands. Presently, CAP (Common Agricultural Policy) recognizes these two land use classes as one, not distinguishing between cultivated and uncultivated grasslands and focusing solemnly on permanent pastures. With the introduction of the hectare payment and the associated rules to vegetation types, some semi-natural grasslands with a fair representation of shrubs, trees and other landscape features will not be eligible for direct payments under Pillar 1.

In addition to agri-environmental schemes, LIFE-the financial instrument for the environment has also been a promoter of grassland conservation projects throughout Europe, mainly in the NATURA 2000 network. The financial support of LIFE projects target various grassland types, and include: preparatory actions, land or rights acquisition, direct conservation actions, monitoring, networking and awareness-raising (EC 2008).Yet, these measures have been questioned on how effective they have been in the conservation of biodiversity (Kleijn and Sutherland 2003). Recently, alternative approaches have been proposed in the conservation policy of traditional farming landscapes (Fischer et al. 2012). Transformation strategy (active management) is a model that links nature to society by first, identifying the needs of the social system, while, managing the social-ecological system as a whole (Fischer et al. 2012). This type of active management through local communities has been applied around the world and is identified as a promising approach to halting biodiversity loss in developing countries where local inhabitants play an essential role in managing and protecting natural resources (Berkes 2004), permitting a reconnection of the social system with nature. Some examples include Mexico, Nepal and Latin America (Bray et al. 2003; Nagendra et al. 2008; Larson and Soto 2008). However, in Fischer et al. (2012) transformation strategy local involvement is only part of the framework, which is strengthened through policy programs rebuilding social links with nature.

Since the implementation of agri-environmental schemes in 1992, the Common Agricultural Policy (CAP) has supported the protection of soil resources throughout European Union. These schemes are targeted to encourage farmers to follow agricultural practices, such as inter alia, no-tillage or countour-tillage, organic fertilization and terracing sloped land. Over the past two decades, the CAP (Common Agricultural Policy) has been gradually reformed towards increasing market orientation. The 2003 reform resulted in the decoupling of direct payments from production and thus regardless if farmer produced commodities or not, farmers were only obliged to keep land in Good Agricultural and Environmental Condition (GAEC). The introduction of these payments led to a reduced pressure of intensive production above market demand but ironically, it is argued by many that the decoupling of direct payments led to extensive abandonment of production and exodus in disadvantaged rural areas (Commission 2003).

The new EU´s CAP reform for 2013, consists of making agricultural economy more competitive by creating jobs, promoting innovation and growth in rural areas while combating climate change. This new direction is targeted towards meeting the EU Biodiversity Strategy for 2020. The target under the new EU Biodiversity Strategy for agriculture consists of *“Maximising areas under agriculture across grasslands, arable land and permanent crops that are covered by biodiversity-related measures under the CAP so as to ensure the conservation of biodiversity and to bring about a measurable improvement in the conservation status of species and habitats that depend on or are affected by agriculture and in the provision of ecosystem services as compared to the EU 2010 Baseline, thus contributing to enhance sustainable management”* (European Commission 2011). Reaching these goals will involve the financial payments, to those playing a part in the conservation of biodiversity. However, these incentives may not be sufficient to halt the decoupling of social and ecological linkages in traditional farming landscapes (Fischer et al. 2012) as much of rural abandonment in mountain areas has already occurred.

1.2. Farmland abandonment: the causes and consequences of landscape change

1.2.1. Land use change and its underlying causes

The socioeconomic and technologic development of human societies has been driving profound changes in the classic relationships between cities and the countryside (Gutman 2007). In the context of such changes, the abandonment of agricultural land is a growing concern throughout much of Europe’s marginal areas, mainly represented in mountainous regions (Agnoletti 2014). Recent trends in land cover/use in Portugal illustrate this phenomenon (Figure 1.2).

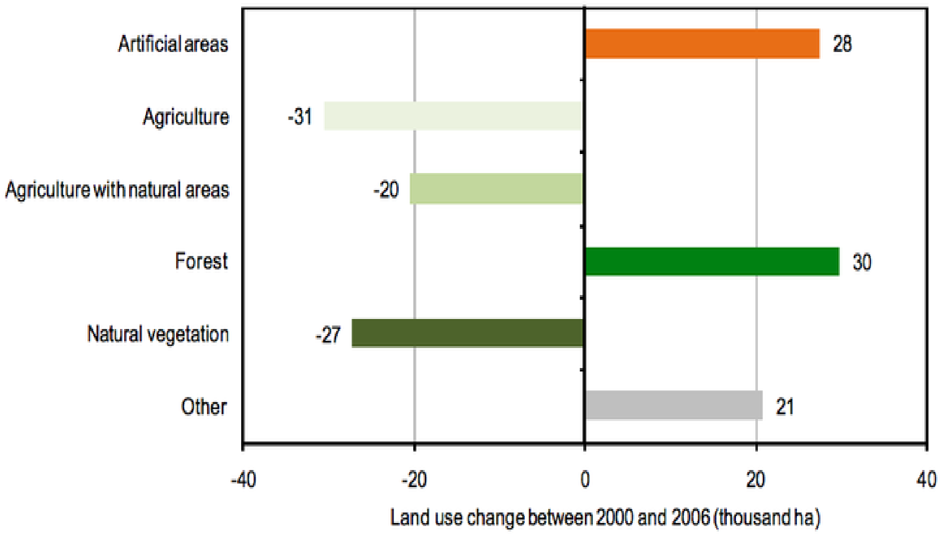

Figure 1.2 Land use change between 2000 and 2006 (Source IGP 2009 and APA 2009).

From a social-ecological perspective, the abandonment of agricultural land represents a change in land use/cover influenced by different drivers. In their review, Rey Benayas et al. (2007) identified three major types of drivers: ecological, socio-economic and un-adapted agricultural systems, and land mismanagement. Ecological drivers include factors such as elevation, fertility, soil depth, climate, among other geo-bio-physical constraints. Socio-economic factors are those related to market incentives, migration of rural populations, technology, industrialization, farmer age, land tenure and accessibility. The last driver is associated to land mismanagement which can lead to soil degradation, flooding and over-exploitation.

Globalization has been identified as one of the main pressures altering these traditional farming landscapes. The occurrence of land abandonment in one area can potentially lead to a shift in agricultural production and land use in other locations, consequently triggering changes in ecological systems (Lambin and Meyfroidt 2011). Over the last decades, we have been witnessing an intensification of land use in more fertile agricultural land and an abandonment of marginal farmlands (Keenleyside and Tucker 2010; Stoate et al. 2009). Of course this complex phenomenon of land abandonment and its patterns are not only dependent on spatial factors but the temporal transformation of the various drivers (Baumann et al. 2011). In Portugal, for example, from 1938 through to 1968, governments implemented the “Plano de Povoamento Florestal”, which involved the plantation of pine on common land, altering local inhabitants’ pasturing systems by losing their rights to any type of management practices

(Aguiar et al. 2009). These and other limitations led to many social and economic problems in rural mountain areas, forcing many to migrate to cities throughout the territory and to other countries, and by the late 1950´s the exodus from rural areas became visible (MacDonald et al. 2000).

In the EU, rural areas represent 57% of the land, and approximately one third of this is considered economically marginal, due to small farm structures and lack of market access (European Union 2010). These limitations and lack of good and services have further contributed to the migration of rural populations to areas of greater opportunities (Rey Benayas et al. 2007; MacDonald et al. 2000). Navarro and Pereira (2012) illustrated the continuous and simultaneous decrease of European agricultural area and rural population in the last 30 years. According to two scenarios (Global Orchestration and Techno-Garden) built by the Millennium Ecosystem Assessment, this trend is expected to continue in the future (Millennium Ecosystem Assessment 2005). However, in most European countries, mass migration has already occurred. According to FAO (2007), approximately 800 million people have migrated to urban centers in the last 50 years.

These changes have not gone unnoticed in political discussions and, in an attempt to halt land abandonment, the CAP designated the Less Favoured Areas (LFAs-Regulation 1257/1999) for which agri-environmental incentives were implemented and funded under axis II of the Rural Development Program. These schemes require farmers to comply with “Good Agricultural Practices” which are approved at national level. LFA´s make up half of Europe´s utilized agricultural areas (MacDonald et al. 2000) and are characterized by their biophysical limitations. Although these incentives were introduced to maintain a viable rural society and promote the sustainability of biodiversity and ecosystem services through good agricultural practices, statistics show otherwise. In the last 40 years, UAAs (Utilised Agricultural Areas) in mountain areas have decreased (MA 2005), promoting ecological succession and encroachment of shrubland and forest, thereby resulting in more homogenous landscapes (Bielsa et al. 2005; Höchtl et al. 2005; Roura-Pascual et al. 2005). According to recent projections, there is an expected increase of abandonment throughout Europe, especially over the next 30 years with lowered levels of CAP support, with Portugal identified as a top abandonment hotspot (Keenlyside and Tucker 2010).

It wasn´t until 1986 that Portugal became a member of the European Union and adopted the EU Common Agricultural Policy (CAP). This altered agricultural systems from a traditional, organic regime to a modern fossil fuel consuming system, mainly in the more productive areas (Baldock et al. 1996). This introduction of technology increased productivity but reduced job opportunities and the exportation of national goods became more difficult due to competition from other EU countries. Another consequence was the lack of capacity of small rural communities, where their agriculture is less productive and economically marginal, to compete with larger productive areas (Baldock et al. 1996).

Drivers of land abandonment can be locally specific, regional/national or global, varying spatially and temporally (Bürgi et al. 2004). In most of Portugal´s rural mountain communities, migration and aging population are the main reasons behind the collapse of traditional farming systems and the increase in land abandonment (Pereira et al. 2005). Together with an intrinsic resistance and incapacity to adopt modern market oriented farming practices, these processes induce consequences (still poorly evaluated) to the environment as well as numerous socioeconomic impacts (Cerqueira et al. 2010; Carvalho-Ribeiro and Lovett 2012; Pereira et al. 2005).

Changes in individual's dependence on local ecosystem services provided by agro-ecosystems have led to an increasing trend of land abandonment (Pereira et al. 2005). Social and demographic drivers, improved access to goods and services and the decline in subsistence agriculture have altered social-ecological dynamics throughout Europe´s rural landscapes (De Aranzabal et al. 2008). These changes have in turn threatened the resilience and sustainability of various ecosystems and their services while contributing to biodiversity change (Busch 2006; Westhoek et al. 2006) and to the potential erosion of traditional agro-ecological knowledge (Cerqueira et al. 2010; Elbakidze and Angelstam 2007), the primary information used in sustaining these systems (Berkes et al. 2000). Although studies have acknowledged the determinants of past agricultural abandonment, few have undertaken the challenge of determining the future economic, ecological and social drivers of farmland abandonment leading to rural out-migration (see Baumann et al. 2011; Keenleyside and Tucker 2010;Benayas et al. 2007). Determining these future drivers has become eminent in formatting regional and local land use planning for rural communities.

Counter-urbanization, the opposite of rural migration, describes a process where people move from urban areas to rural areas, and has been the focus of past studies (Hoggart 1997; Jones et al. 1986), primarily focused on its economic impacts (Stockdale et al. 2000; Findlay et al. 2000). Many of the factors behind counter-urbanization can be applied to rural out-migration. Presently, many of the studies pertaining to rural out-migration have focused on four main aspects. Youth out-migration has been one of the major focuses of rural migration studies, and is driven mainly by the lack of employment opportunities and communities, low manual and technical needs (Machold et al. 2002). In some regions throughout Europe we find the opposite trend, where individuals prefer to remain in the countryside (Portela and Gerry 2002), which can be connected to the individuals' attachment to the region. Other studies have focused on determining the physical, demographic and development impact of this phenomenon. The aging of the local population has also been a consequence of this selective out-flow (Stasiak 1992). These demographic changes have resulted in declines of the communities' available services, such as schools and health centers (Pereira et al. 2005).

Out-migration is thought to bring positive benefits to those who have chosen to leave. Still, it has been argued that those who migrate have a fairly low educational level, and generally hold low paying jobs (Kasimis et al. 2003). In London, a recent study showed that many homeless youths migrated from rural areas, suggesting that migration does not solve all economic challenges (Stockdale 2004). And may in fact lead to social changes and a feeling of disconnect with their home community. This is not the case however for all regions. For example, in Portugal many families who have chosen to migrate frequently return, contributing to the local economy. According to some authors, we need to examine counter-urbanization at different geographical scales (Vartiainen 1989). For example, we can have two different trends occurring simultaneously, local depopulation can persist within remoter regions that are experiencing repopulations.

1.2.2. Human well-being and its connection to landscape change

According to the Millennium Ecosystem Assessment (MA), human well-being is defined by the following dimensions: material minimum for a good life, health, good social relations, security, and freedom of choice (MA 2005). In this context, ecosystem services relate the benefits that humans obtain from the various ecosystem functions, contributing to human well-being (Costanza et al. 1997; MA 2005). Changes in the provision of ecosystem services will affect human well-being, impacting its various determinants, influenced by each individual's freedom of choice (Figure 1.3).

As part of Portugal´s Sub-Global Assessment, a study carried out in a rural mountain community revealed that in the last 50 years some determinants of human well-being had improved, such as material and security. However, others such as social well-being had decreased along with the dependence of local ecosystem services

(Pereira et al. 2005). In addition to these changes, local residents attribute a negative connotation to traditional agro-ecosystems.

These changes are mainly due to economic development throughout the 20th century, bringing subsistence agricultural lifestyles in marginal mountain areas to a disadvantage, leading to a decrease in job opportunities, and thereby promoting rural exodus (Benayas et al. 2007; Khanal and Watanabe, 2006; Lasanta et al. 2005; MacDonald et al. 2000). These socio-economic and political drivers have contributed to changes in rural inhabitants' perception of value (Calvo-Iglesias et al. 2009; Nikodemus et al. 2005; Roura-Pascual et al 2005; Westhoek et al., 2006) and led to local people's disconnection or lack of dependence on subsistence farming (Aguiar et al. 2009; MacDonald et al. 2000), modifying the patterns of land use and land management (Calvo-Iglesias et al. 2009; Lasanta-Martínez et al. 2005; Plieninger et al. 2006).

The task of documenting human's perception, preference or value of landscapes is complex and multi-level (Sayadi et al. 2009; Soini and Aakkula 2007; Soliva and Hunziker 2009). However, incorporating people´s perception into landscape planning is indispensible. According to the European Landscape Convention (ELC), "landscape" means an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors (European Council 2000). The aim of ELC is to encourage peoples and governments throughout Europe to care for all landscapes (not only those landscapes such as National Parks or UNESCO World Heritage Landscapes), through adequate identification, assessment, protection, management and planning. Presently, 40 countries have signed the European Landscape Convention, including Portugal. Ultimately, determining human appreciation for a particular landscape helps delineate socially accepted management and planning, reducing conflict among stakeholders and improving the quality of decision-making. Erv Zube (1931-2002) defined landscape planning as an activity concerned with reconciling competing land uses while protecting natural processes and significant cultural and natural resources. Yet, the way an area of land is perceived by people many not run parallel to the region's full ecological importance.

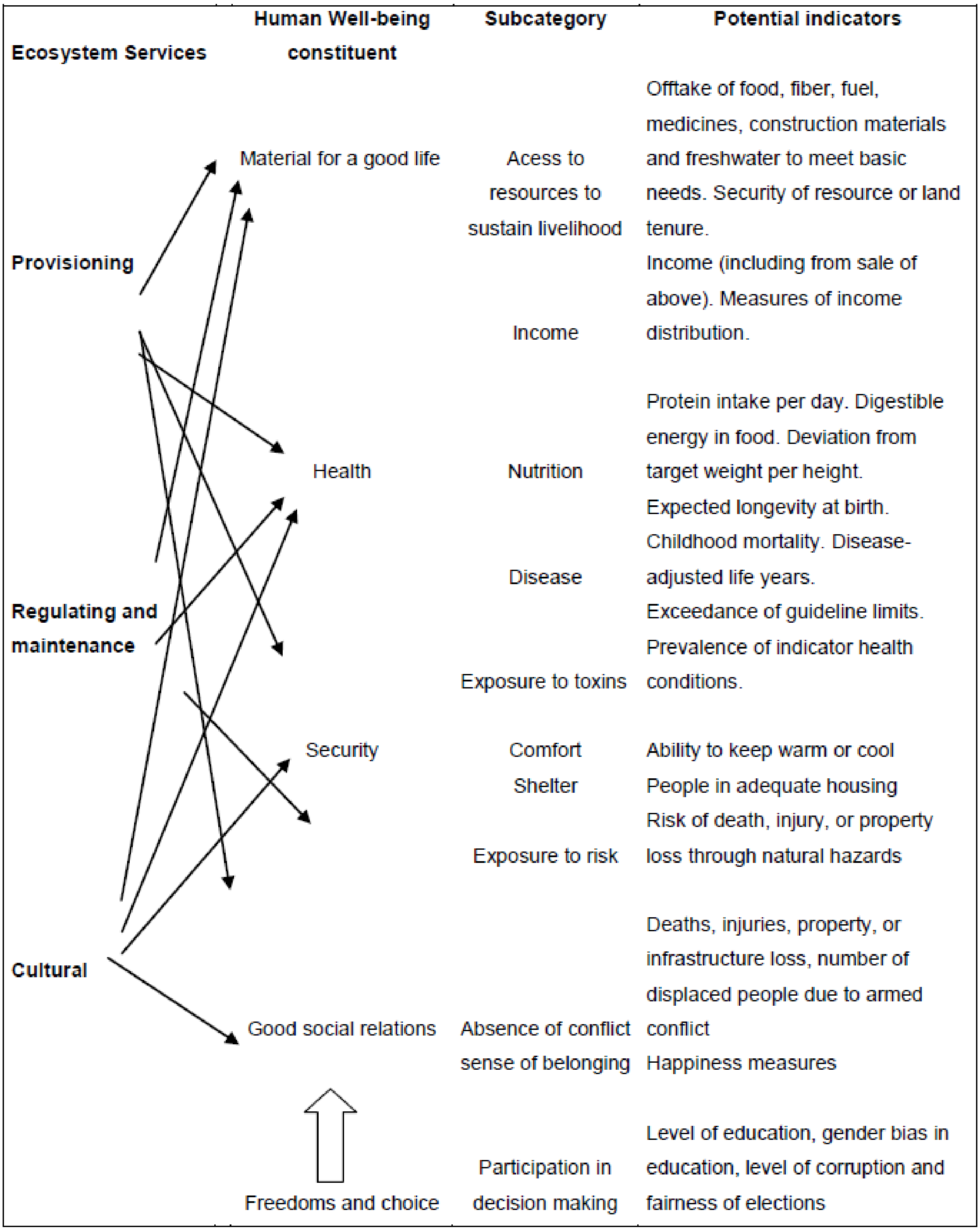

Figure 1.3 The constituents of human well-being and the connection with ecosystem services, recognized by the Millennium Ecosystem Assessment (MA 2005). Potential indicators of human well-being have also been listed.

The way humans perceive a landscape plays a fundamental role in landscape change, and humans are frequently modifying their landscape for ecological, social or economic benefits (Gobster et al. 2007). The most common conceptualization of human-landscape relationship identifies humans as the primary mediator of change to the landscape. Humans induce changes at various scales, intensity and type, e.g. land conversion from forest to agriculture, or from agricultural area to abandoned farmland. Human-landscape relationships have also been defined by viewing humans as recipients of information from the landscape. This conceptualization of the human-landscape relationship is developed in the perceptual and cognitive branches of psychology. This form of relationship is most commonly used in the field of landscape appreciation, specifically in the area of aesthetic perception (Zube et al. 1982). In this context, human preference for a particular landscape is mainly emotionally driven, where elements of the landscape play an important role in perception. In the conceptual model of human-environmental interactions by Gobster et al. (2007), ecological aesthetics affects landscape planning, design and management. The basis of their conceptual model of human-environmental interactions is defined by the critical perceptible realm (surrounding landscape), the scale at which aesthetic experiences occur and directly or indirectly drive landscape changes affecting ecological processes and their capacity to provide ecosystem services (Figure 1.4). Their model also proposes that landscape change impacts aesthetic experience and aesthetic experience benefits human well-being. Zube (1987) describes a similar human-landscape relationship defined as a continuing transactional process. This concept is built on environmental psychology theories, which describe the perception and changes to the landscape as a function of the transactions. As Zube et al. (1982) note:

> “The human component encompasses past experience, knowledge, expectations and the socio-cultural context of individuals and groups. The landscape component includes both individual elements and landscape as entities. The interaction results in outcomes which in turn affect both the human and landscape components.”

The various conceptualizations of the human-landscape relationship exemplify how human perception influence landscape change and vice-versa. There are, however, other theories which focus on the philosophical and ethical values of how humans value nature. The complexity of these issues will not be discussed further in this thesis.

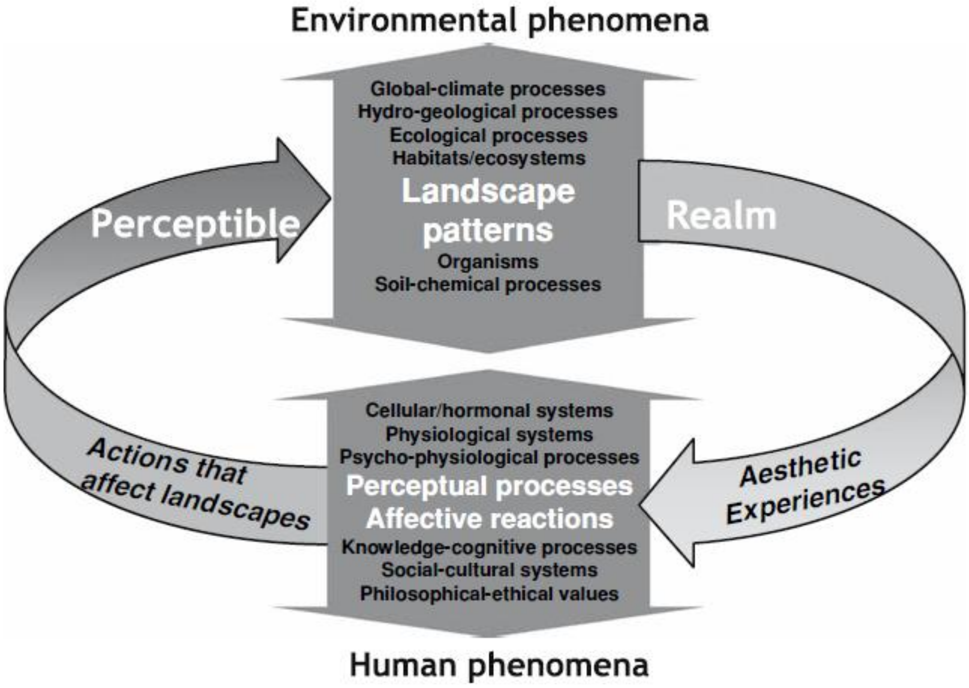

Figure 1.4 Model of human-environmental interactions in the landscape (Gobster et al. 2007).

Documenting local inhabitants' perception of changes and preferences provides different perspectives for better decision making and trade-offs in the management of services. Several studies have identified landscape preference according to socio-demographic differences (Natori and Chenoweth, 2008, Soliva and Hunziker 2009); others have focused on the theory of place attachment (Walker and Ryan 2008). Several others have established a relationship between landscape preference and environmental value orientations (Kaltenborn and Bjerke, 2002; Van den Berg and Koole 2006). Finally, others have studied preference in reference to landscape change (Hunziker et al. 2008) and land uses (Benjamin et al. 2007). Although many studies have addressed issues related to preference and perception of landscape, very few studies have determined preferences for different ecosystems and their services using a non-economic valuation approach (Martín-López et al. 2012). The recognition of the existence of diverse perceptions of value and preference for a given natural resource by different stakeholders is indispensable and imperative in future decision making in land use planning and in outlining conservation projects supporting sustainability goals (Bassi and Tache 2011; Swiderska 2003; Sayadi et al. 2009; Soliva and Hunziker 2009).

1.3. Ecosystem services: the future backbone of mountain rural landscapes?

1.3.1. Ecosystem services: conceptual framework and trends

Ecosystem services are defined as the benefits humans derive directly or indirectly from ecosystems (Daily 1997; MA 2005). The Millennium Ecosystem Assessment (MA) was the largest study ever made linking ecosystems to human well-being. It was the stepping stone in providing a framework for assessing ecosystems and classifying their services into four groups: supporting, provisioning, regulating and cultural services. Supporting services are necessary for the production of all other ecosystem services and include nutrient dispersal and cycling, seed dispersal and primary production. Provisioning services are those services obtained from ecosystems and include food (i.e. seafood and game) crops, wild foods and spices, water, minerals, pharmaceuticals, biochemicals and industrial products and energy. Regulating services are benefits obtained from the regulation of ecosystem processes, such as carbon sequestration and climate regulation, waste decomposition, purification of water and air, crop pollination, pest and disease control. Cultural services are non-material benefits obtained through spiritual, recreation and aesthetic experiences.

Since the publication of the MA, several upgraded versions of classifying ecosystem services have been developed. The most recent and widely used include the TEEB (The Economics of Ecosystems and Biodiversity) and the CICES (Common International Classification of Ecosystem Services) typologies. A synthesis of these two international classification systems can be found in Maes et al. (2013). Under the most recent conceptual framework of ecosystem services developed by the European Environmental Agency (EEA), the CICES, there are only three groups of services: provisioning, regulation and maintenance, and cultural (Table 1.1).

The CICES system has been adopted as typology for ecosystem services under the ecosystem assessments that are requested under Action 5 of the EU biodiversity strategy to 2020, which asks the EU member states to map and assess the state of Europe’s ecosystems and their services by 2014, and to assess their economic value by 2020. The conceptual framework for the assessment under Action 5 is illustrated in Figure 1.5. Under this classification, ecosystem services are defined as the contributions that ecosystems make to human well-being, and where biodiversity is the primary provider of basic ecosystem function and processes, linking socio-economic systems with ecosystems through the flow of services, directly

Table 1.1 The Common International Classification of Ecosystem Services (CICES; Haines-Young et al. 2009).

Theme	Class	Group
Provisioning	Nutrition	Terrestrial plant and animal food
		Freshwater plant and animal food
		Marine plant and animal foodstuffs
		Potable water
	Materials	Abiotic materials
		Biotic materials
	Energy	Renewable biofuels
		Renewable abiotic energy sources
Regulation and Maintenance	Regulation of wastes	Bioremediation
		Dilution and sequestration
	Flow regulation	Air flow regulation
		Water flow regulation
		Mass flow regulation
	Regulation of physical environment	Atmospheric regulation
		Water quality regulation
		Pedogenesis and soil quality regulation
	Regulation of biotic environment	Lifecycle maintenance and habitat protection
		Pest and disease control
		Gene pool protection
Cultural	Intellectual and experiential	Recreation and community activities
		Information and knowledge
	Symbolic	Religious and spiritual
		Aesthetic, heritage

consumed or enjoyed by humans (Maes et al. 2013). Biodiversity plays a key role in the structural set-up of ecosystems which is essential to maintaining basic ecosystem processes and supporting ecosystem functions. The state of ecosystems is specifically addressed in the framework. The argument is that healthy ecosystems (in good status) possess the full potential of ecosystem functions (Maes et al. 2013).

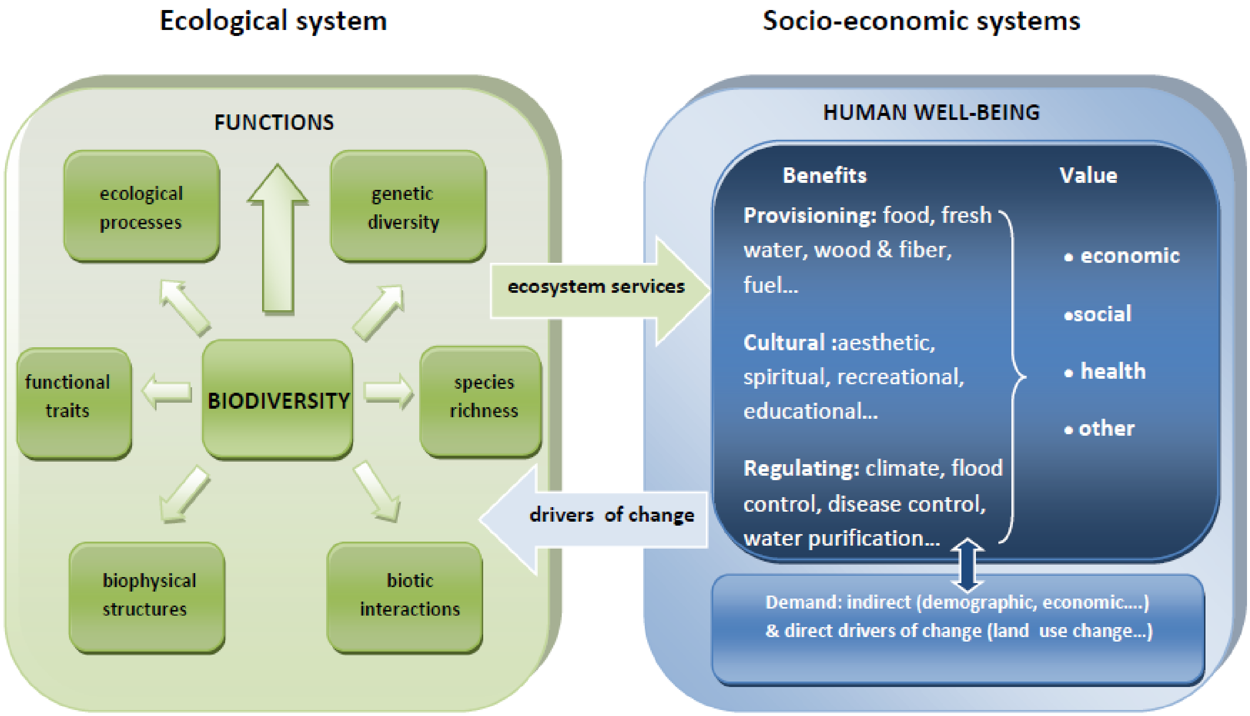

Figure 1.5 Conceptual framework of ecosystem services (adapted from Maes et al. 2013).

Ecosystem services play a fundamental role in human well-being (Figure 1.5) and according to the MA human use of ecosystem services are increasing, corresponding with the growth in earth´s human population. However, the demand for services such as wood fuel, agricultural fibers, wild terrestrial foods and wild caught fish have decreased (MA 2005).

Degradation and loss of habitats, overexploitation, population increase, species invasion and climate change have all contributed to the current depleted supply of ecosystem services (MA 2005). Land use change has been identified as one of the primary drivers of ecosystem service decrease and considerable attention has been given to how land use changes have impacted ecosystems and the provision of ecosystem services (Schrötter et al. 2005). It is estimated that approximately 60% of ecosystem services are globally being degraded or used unsustainably (MA 2005).

Today, policy makers, conservationists and economists face the challenge of increasing the sustainability of the supply of ecosystem services while maintaining environmental, social and economic benefits. Ecosystem services are provided by both natural habitats as well as by human modified systems such as agricultural ecosystems. However, centuries of agricultural activity have resulted in the sacrifice of a number of ecosystem services, mainly regulating and maintenance services (MA 2005). The on-going land abandonment, felt over the last 50 years throughout much of Europe´s rural mountains, may have created an opportunity for those services once suppressed at the hand of Man. Forests, for example, faced considerable reduction in the past, due to agricultural activity. Today, they make up 41% of Europe´s mountains, providing a high number of services such as carbon sequestration, erosion prevention, air quality regulation, recreation, regulation of soil and water, among others (Maes et al. 2012a; 2012b). In particular, they are regulators of natural disaster, as soils have high infiltration rate reducing peak flows and floods (Maes et al. 2009). These services directly or indirectly contribute to human well-being while holding economic and ecological benefits. The ambitious new EU biodiversity and ecosystem targets have exerted a further pressure in mainstreaming ecosystem services into policy decisions (European Commission 2011). This implies looking at both the positive and negative effects policies can have on ecosystems and the services supplied and at how changes to policies can contribute to human well-being through the promotion of ecosystem services. Ideally, the perfect model would be to maximize both biodiversity and ecosystem services; however, this is often impossible and consequently policies need to prioritize their goals based on assessments (Maes et al. 2012a). This requires spatially explicit data and models. The PRESS (PEER Research on Ecosystem Services) initiative provided data on the various methods for mapping services, assess and value ecosystem services at various scales. One of their primary conclusions was the actual greening of farmers’ subsidies in Europe, improving water quality and consequently leading to both social and economic benefits (Maes et al. 2013).

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1.3.2. European mountains: their people, ecosystem services and future prospect

Mountains are the ecological backbone of Europe, covering 35% of the territory and supplying a wide range of ecosystem services (Harrison et al. 2010). However, it wasn't until the early 1990´s that mountain areas were given the needed attention. Globally, they have been considered in the UN Framework Convention on Climate Change, in a Programme of Work for Mountain Diversity under the Convention on Biological Diversity (2004) and in the Millennium Ecosystem Assessment (Körner and Ohsawa 2005).

Funded by the European Commission as a 6th Framework Coordination Action Project, the RUBICODE project had the main objective of assessing the status and trends of ecosystem services in Europe, previously evaluated by the MA at the global scale (Körner et al. 2005). The RUBICODE project provided a ranking of importance of ecosystem services supplied by European mountain systems (Table 1.2). Some of these services include climate regulation, air quality, controlling natural hazards, tourism, recreation, and aesthetic appreciation (Harrison et al. 2010). In particular, mountains are referred to as "water towers", supplying water to more than half the human population. Therefore, the supply of mountain ecosystem services is beneficial not only for local communities but also for residents in lowlands (Körner et al. 2005).

Mountain ecosystems are intrinsically multi-functional, however human use and status of different ecosystem services vary across mountain regions in Europe (see Harrison et al. 2010). But to plan for multifunctional mountains, researchers need to determine which ecosystem services are supplied, and where, as well as their human demand and value. From Table 1.2 we can see that for some services, mainly regulating (e.g. pollination and disease regulation), information is still insufficient, complicating their valuing (Harrison et al. 2010).

Table 1.2 Ranking of importance of ecosystem services supplied by European mountain systems (Source: Haslett 2010, adapted from Harrison et al. 2010).

MA category	Ecosystem service	Key contribution	Some contribution	No contribution	Poorly known
Provisioning services	Food and fibre	X			
	Timber/fuel/energy	X			
	Freshwater	X			
	Ornamental resources		X		
	Biochemicals/natural medicines		X		X
	Genetic resources	X			X
Regulating services	Pollination		X		X
	Seed dispersal		X		X
	Pest regulation		X		X
	Disease regulation		X		X
	Invasion resistance				X
	Climate regulation	X			
	Air quality regulation	X			
	Erosion regulation	X			
	Natural hazard regulation	X			
	Water flow regulation	X			
	Water purification/waste treatment		X		X
Cultural services	Spiritual and religious values		X		
	Education and inspiration	X			
	Recreation and ecotourism	X			
	Cultural heritage	X			
	Aesthetic values	X			
	Sense of place	X			

Roughly 17% (118 million) of Europe´s population live in mountain areas. The low population densities in mountain systems are mainly associated to their harsh living conditions (i.e. physical constrains). However, economic and social drivers have been the primary factors influencing recent population trends and land abandonment (Keenleyside and Tucker 2010). In some mountain regions of Poland, Serbia, Slovenia and Switzerland, population numbers have gone up, in contrast to Finland, Italy, Portugal and Sweden (Haslett 2010). Generally, mountainous farmland areas are not economically viable, due to land restrictions. In fact, 95% of all mountain systems are designated as Less Favored Areas (LFAs-Regulation 1257/1999) a measure implemented by the European Common Agricultural Policy whose objective was to halt rural abandonment and maintain agricultural activity through the promotion of incentives (Dax 2005; Stoate et al. 2009). Yet, till date this measure has not been successful at halting rural exodus and consequently, land abandonment (Navarro and Pereira 2012).

Farmers have played an essential role in maintaining these landscapes, but many mountain systems are now being abandoned. While seen by many as a global negative impact on biodiversity, others see it has an opportunity for the “lost nature” and the promotion of ecosystem services (Navarro and Pereira). As the aim to halt the loss of biodiversity until 2010 set by the Convention on Biological Diversity has not been met (European Commission 2011), a new strategy has been defined. The New Biodiversity Strategy for 2011-2020 adopted by the European Commission states that all countries and stakeholders should halt biodiversity and degradation of ecosystem services by 2020, restore ecosystems and avert global biodiversity loss (European Commission 2011). This new biodiversity strategy goes hand in hand with a fairly recent initiative, “Rewilding Europe”, launched by several entities and aimed to rewild one million hectares of land by 2020. The emergence of this topic is seen as a promising solution to Europe´s on-going rural abandonment. The restoration of Europe´s abandoned land is thought to provide humans with economic benefits through the promotion and conservation of ecosystem services generated from wilderness or areas of high degree of naturalness (Rey Benayas et al. 2009).

High wilderness representation is mainly associated to mountain systems (Price 2010). Nordic mountains embody the largest proportion (28%) of wilderness areas, followed by the Pyrenees (12%), eastern Mediterranean islands and Alps (9%), and British Isles (8%). Wild ecosystems are healthy systems that provide a wide range of ecosystem services. They are stable and self-sustainable, able to maintain their structure, function and resilience over time (Costanza and Mageau 1999). They play an important role in protecting services such as the air we breathe, the water we drink and the wildlife we value, such as the existence of charismatic species, which include the bison and the bear among other species that are reliant on wilderness areas (Russo 2006). Wild ecosystems also have the capacity to supply higher levels of services. For example, there is higher carbon storage capacity in undisturbed forest, peatland and wetland (Schils et al. 2008).

At the European-level, the “Pan Parks” initiative is a network of wilderness reserves with the aim of protecting and managing wilderness areas as well as promoting sustainable tourism in these areas. It now consists of 13 national parks in eleven countries. Some examples include the Majella and the Retezat National Parks, in Italy and Romania respectively, where the reintroduction and conservation of large carnivores and ungulates have contributed to the local economy. In particular, the “Safeguarding the Romanian Carpathians Ecological Network” is a successful case study on rewilding initiatives and has focused on safeguarding the natural capital through a green economy boosting local revenues (Maanen et al. 2006).

The “Natura 2000” is a network of protected areas, targeted at the conservation of biodiversity while providing a set of benefits to society and the economy through ecosystem services. Currently, a total of 43% of Natura 2000 sites are in mountain areas and 29% for the EU as a whole (ten Brink et al. 2011). The Iberian mountains hold the highest Natura 2000 coverage, followed by Slovenia, Slovakia and Spain. The Natura 2000 network aims to protect the most valuable and threatened species and habitats in Europe through the designation of Special Areas of Conservation (SACs), Habitats Directive (92/43/EEC) (EC, 1992) and Special Protection Areas (SPAs) under the Birds Directive (79/409/EEC) (EC 1979). Mountain systems are home to 42 of the 231 habitat types listed in Annex I of the Habitats Directive (92/43/EEC). For example, most of natural grasslands habitat types are limited to mountain areas. Moreover, the proportion of habitat with favourable status is high in comparison to other regions (Halada et al. 2011).

The proportion of total area of Natura 2000 sites in mountains is not uniform across Europe. Cyprus (95%), Slovenia (83%), Greece (82%), and Italy (81%) were the top countries whose mountain systems are within the Natura 2000 network (Figure 1.6). For these countries, roughly all have half of their national area represented by mountain areas. Conversely, in countries such as Belgium (8%), Finland (9%) and Luxembourg (11%), which have less than 20% of total area of Natura 2000 in mountain areas and generally, mountain systems cover less than 10% of the territory (Figure 1.6).

Over the years, the recognition that protected areas potentially provide socio-economic benefits has resulted in several valuation studies under the TEEB initiative (TEEB 2010; TEEB 2011). These studies evaluated the costs of the loss of biodiversity and the decline on ecosystem services, comparing these costs to those of effective conservation and sustainable use. The Natura 2000 network exemplifies a cost effective means of protecting wildlife while generating benefits. Annually, the gross socio-economic and co-benefits (social and environmental) from the Natura 2000 network range between €223 billion and €314 billion, representing between 2 and 3% of EU´s GDP (ten Brink et al. 2011). This contrasts with the annual implementation cost estimated at € 5.8 billion for the EU-27 (Gantioler et al. 2010) while providing 8 million (FTE) jobs (BIO Intelligence Service, 2011). The TEEB country case-studies present further evidence for the economic value of forests, freshwater ecosystems, soils and coral reefs as well as the social and economic costs of their loss (TEEB 2012).

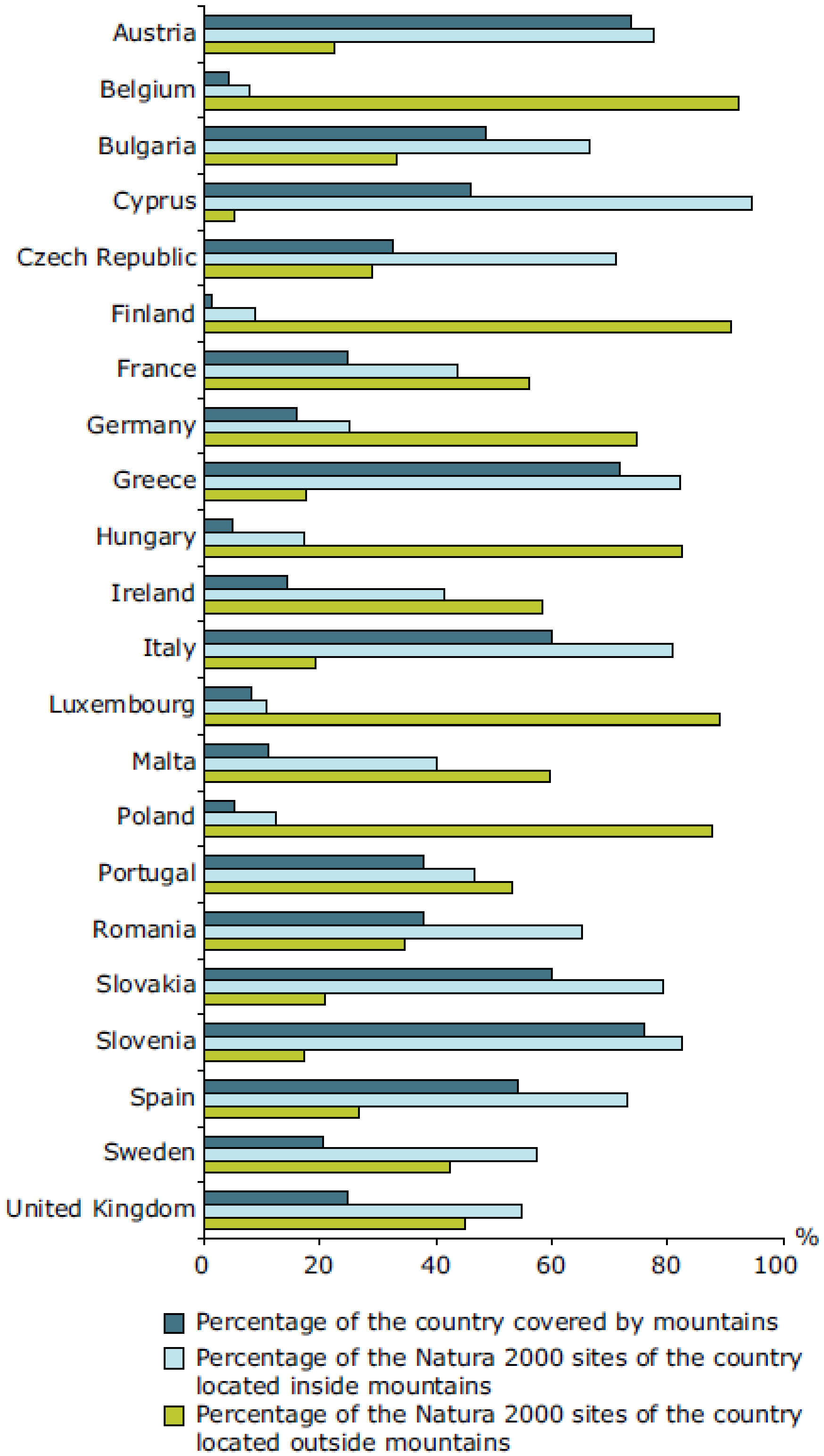

Figure 1.6 Percentage of area covered by Natura 2000 sites inside and outside mountains by country, and of area covered by mountains (Source: Price 2010).

Managing ecosystems through ecosystem services is feasible, guaranteeing the flow of services and generating socio-economic benefits (Price 2010). Presently, there is a new economy being developed around carbon storage and payments for ecosystem services, representing potential income to rural areas. The restoration of Europe´s natural landscape can help create a connecting network of functioning ecosystems, increasing biodiversity and the capacity of ecosystems to provide services (Rey Benayas et al. 2009).

1.4. Research objectives and thesis overview

1.4.1. General context and overarching research goals

Abandonment of Europe´s low intensity farming has caused major landscape transformation, particularly in rural mountain systems (Anthrop 2005). The preservation of these traditional agricultural landscapes has been a major goal in agri-environmental schemes and other policy measures. Still, the abandonment of farmland has continued, driven mainly by socio-economic factors (Benayas et al. 2007). Farmland abandonment is considered one of the major land use changes impacting biodiversity and ecosystem services, and consequently human well-being. However, at the European scale, this phenomenon is also seen as an opportunity for wildlife and wild areas, which have along the years been sacrificed due to human use of the land (Höchtl et al. 2005).

Although land abandonment is not a recent phenomenon throughout Portugal´s rural mountain communities, very few information can be found on the social perception of these changes and the impacts they have had on ecosystem services, biodiversity and human well-being. In future projections for 2030, Portugal was identified as a major hotspot for land abandonment (Keenleyside and Tucker 2010). One of the main factors contributing to Portugal´s overrepresentation of farmland abandonment is the location of the country's agricultural areas and the lack of intergenerational succession (Bernúes et al. 2011). Presently, 86% of total agricultural area (UAA) is designated as Less Favoured Areas (LFA). In Europe, LFA have increased from a third in 1975, to more than half in 2005 (Dax 2005; MacDonald et al. 2000). Furthermore, in the EU-27, 55% of farm holdings are managed by farmers with 60 years of age and older, and in Portugal this number rises to 73.4% (EUROSTAT, FSS, 2003-2007). In this context, it is of key importance to document and understand how landscapes and human influence have changed along the years, to effectively develop future policies and management plans which promote ecological, social and economic benefits, to both rural communities and overall society.

Yet, the abandonment of marginal areas across Europe has opened a window of opportunity for the restoration of nature. Rewilding, a fairly recent concept in Europe, has been questioned on its efficiency and capacity to sustain biodiversity and ecosystem services and ultimately contribute to human well-being. Although we are still unaware of the full impacts land abandonment will have on food security, rewilding is seen as an opportunity to bring back what has been lost due to centuries of anthropogenic intervention.

In this context, the research developed for this thesis aimed: (1) to determine human perception of landscape change and preference as well as of ecosystem services along an urban-rural gradient; (2) to analyze the spatial changes of land use and the drivers behind abandonment, as well as the impacts it has had on ecosystem services and biodiversity in mountain rural areas; and (3) to determine the ecological benefits of rewilding landscapes undergoing abandonment in Europe.

1.4.2. Specific objectives and research questions

The research presented in this thesis was thus organized around three main objectives, each one assessed according to specific research questions.

1) *To identify divergent preferences and perception concerning landscape change and ecosystem services along an urban-rural gradient*, by addressing the following questions:

- **a)** Do local residents' preference for ecosystems and their services, as well as the determinants defining human well-being, vary across an urban-rural gradient? How do local residents perceive their surrounding landscapes today and in the future, and what is their opinion of future conservation efforts?
- **b)** Do residents perceive landscape change? Are the changes referred by those residents consistent with scientific assessment based on land cover data?

2) *To identify the key drivers and the main consequences of land abandonment*, by addressing the following questions:

- **a)** What are the underlying drivers of agro-pastoral abandonment? Have those drivers fluctuated through time?
- **b)** Will current residents consider leaving their rural villages? If so, what are the determinants (and thresholds) of locals willing to leave?
- **c)** What are the gains and losses among ecosystem services in land which has been undergoing abandonment for the last five decades?

3) *To identify the social-ecological benefits of rewilding landscapes across Europe*, by addressing the following questions:

- **a)** Does the supply of ecosystem services vary across a gradient of wilderness quality? In particular, do high quality wilderness areas provide a greater diversity of ecosystem services?
- **b)** Do potential rewilded areas supply high quality ecosystem services? More specifically, will future abandoned land provide more ecosystem services than present land use (agriculture)?

1.4.3. Description of the thesis outline

The thesis is organized in seven chapters. Each chapter (except for chapters 1, 5 and 7; see below), is then organized into six sections: an Introduction section, in which we describe the context and objectives of the research performed; a Methods section, in which we describe the study areas as well as the main databases and methodological frameworks; a Results section, where we describe and interpret the main findings of our research; a Discussion section, in which we discuss the main results described before; a Conclusion section, in which we outline the main conclusions; and finally a list of References.

In Chapter 1 (the present chapter), we provided a general introduction to the research field, providing a detailed state of the art review. It includes a conceptual framework relating land abandonment to human aesthetic perception, the drivers of land abandonment and the importance of ecosystem services as our primary life supporting system. Subsequently, we outlined the overarching research goals and the specific research objectives related to each case study.

Chapters 2 through 5 comprise different research goals and describe the research performed to address them. Chapter 2 focuses on determining the present and potential future drivers of land abandonment and rural migration. Specifically, we determined each individual’s threshold based on the economic utility of farmland, using the model of Figueiredo and Pereira (2011) on the economics of migration to support our data. We hypothesize that those individuals who are able to maintain agricultural utility are less willing to leave.

Chapter 3 focuses on the human preference for ecosystem types and for key ecosystem services, as well as perception of landscape change along an urban-rural gradient. We also identified local residents’ preference for future conservation efforts and the elements of conservation value. To determine the existence of significant differences in perception and preference, a total of 40 individuals were questioned for each rurality class (urban-rural, transition rural, and deep rural). The influence of a rurality gradient and conservation perspectives were also analyzed.

In Chapter 4, we spatially and temporally quantify the gains and losses in potential ecosystem service supply and in biodiversity spanning a 50 year period of mountain farmland undergoing abandonment. We focus on three ecosystem services: carbon sequestration and storage, sediment retention and water purification, and changes to biodiversity. To do so we apply the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) software system (Natural Capital) to model three services: carbon storage and sequestration, sediment retention, and water purification / nutrient retention. For biodiversity, we used a multi-habitat species-area relationship (SAR) developed by Pereira and Daily (2006), and applied in the study area by Proença and Pereira (2013) and by Guilherme and Pereira (2013).

In Chapter 5 we investigate the supply and spatial distribution of ecosystem services at a pan-European scale and particularly in wilderness areas. We then use the supply of ecosystem services in wilderness areas as a proxy for areas that are projected to undergo land abandonment and rewilding in the future. Lastly, we discuss the various economic and ecological benefits of rewilding in Europe.

To conclude, in Chapter 6 we summarize the key findings, present the main conclusions, and outline directions for future research.

1.5. References

Agnoletti, M. (2014).Rural landscape, nature conservation and culture: Some notes on research trends and management approaches from a (southern) European perspective. Landscape and Urban Planning.

Aguiar, C., Rodrigues, O., Azevedo, J., Domingos, T. (2009). Montanha. In H. M.

Pereira, T. Domingos, L. Vicente, V. Proença (Eds), Ecossistemas e Bem-estar Humano: *Avaliação para Portugal do Millenium Ecosystem Assessment* (pp. 295-339). Fundação da Faculdade de Ciências da U.L e Escolar Editora.

Angelstam, P., Elbakidze, M., Axelsson, R., Lopatin, E., Sandström, C., Törnblom, J., Dixelius, M., Gorchakov, V. & Kovriga L. (2007). Learning for sustainable forest management: Europe's East and West as a landscape laboratory. *Forest facts*, 1(2007), Swedish University of Agricultural Sciences.

Antrop, M. (2005). Why landscapes of the past are important for the future. *Rural Landscapes: Past Processes and Future Strategies*, *70*(1–2), 21–34. doi:10.1016/j.landurbplan.2003.10.002

APA (2009). Relatório do Estado do Ambiente. Agência Portuguesa do Ambiente. Amadora

Andersen E (ed.) (2003) Developing a high nature value farming area indicator. Internal report. EEA, Copenhagen, 76 pp

Baldock D, Beaufoy G, Bennett G, Clark J (1993) Nature conservation and new directions in the common agricultural policy. Institute for European Enviromental Policy London, Arnhem

Baldock, D., Beaufoy, G., Brouwer, F., Godeschalk, F. (1996). Farming at the Margins: Abandonment or Redeployment of Agricultural Land in Europe. Institute for European and Environmental Policy and Agricultural Economics Research Institute, London.

Bassi, M. and Tache, B. (2011). The community conserved landscaped of the Borana Oromo, Ethiopia. Management of Environmental Quality 22, pp. 174-186.

Baumann, M. et al (2011) Patterns and drivers of post-socialist farmland abandonment in Western Ukraine Land Use Policy, 28:552–62

Benayas, J. M. R., Newton, A. C., Diaz, A., & Bullock, J. M. (2009). Enhancement of Biodiversity and Ecosystem Services by Ecological Restoration: A Meta-Analysis. *Science*, *325*(5944), 1121–1124.

Benayas, J. M., Martins, A., Nicolau, J. M. and Shulz, J. J. (2007). Abandonment of agricultural land: an overview of drivers and consequences. CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources, 2(57):1-14.

Benjamin, K., Bouchard, A., & Domon, G. (2007). Abandoned farmlands as components of rural landscapes: An analysis of perceptions and representations. *Landscape and Urban Planning*, *83*(4), 228–244. doi:10.1016/j.landurbplan.2007.04.009

Berkes, F. (2004). Rethinking Community-Based Conservation. *Conservation Biology*, *18*(3), 621–630. doi:10.1111/j.1523-1739.2004.00077.x

Bernués, A., Rodríguez-Ortega, T., Ripoll-Bosch, R., Casasús, I. 2011. A qualitative esearch on Spanish farmers and citizens perceptions of ecosystem services provided by mountain livestock farming. 17th Meeting of the FAO-CIHEAM Mountain Pasture Network. Pastoralism and ecosystem conservation 5-6-7 June 2013 – Trivero – Italy, ORAL PRESENTATIONS 12

Bielsa, I., Pons, X., & Bunce, B. (2005). Agricultural Abandonment in the North Eastern Iberian Peninsula: The Use of Basic Landscape Metrics to Support Planning. *Journal of Environmental Planning and Management*, *48*(1), 85–102. doi:10.1080/0964056042000308166

Bürgi, M., Hersperger, A., & Schneeberger, N. (2004). Driving forces of landscape change — current and new directions. *Landscape Ecology*, *19*(8), 857–868. doi:10.1007/s10980-004-0245-8

Calvo-Iglesias, M. S., Crecente-Maseda, R., & Fra-Paleo, U. (2006). Exploring farmer's knowledge as a source of information on past and present cultural landscapes: A case study from NW Spain. *Landscape and Urban Planning*, *78*(4), 334–343. doi:10.1016/j.landurbplan.2005.11.003

Calvo-Iglesias, M. S., Fra-Paleo, U., & Diaz-Varela, R. A. (2009). Changes in farming system and population as drivers of land cover and landscape dynamics: The case of enclosed and semi-openfield systems in Northern Galicia (Spain). *Landscape and Urban Planning*, *90*(3–4), 168–177. doi:10.1016/j.landurbplan.2008.10.025

Carvalho-Ribeiro, S.M., Lovett, A. (2012). Identifying assets and constraints for rural development with qualitative research tools: A case study of Castro Laboreiro, Portugal. Landscape and Urban Planning.

Cerqueira, Y. C., Araújo, C., Vincente, J., Pereira, H. M. and Honrado, J., 2010. Ecological and cultural consequences of agricultural abandonment in the Peneda-Gerês National Park (Portugal). In: T. Evelpidou, T. Figueiredo, F. Mauro, V. Tecim, A. Vassilopoulos. (Eds), *Natural Heritage from East to West: Case studies from 6 EU countries* (pp. 175-183). London, New York.

Costanza, R., d'Arge, R., de-Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neil, R., Paruelo, J., Raskin, R., Sutton, P., van den Belt, J., (1997). The value of the worlds ecosystem services and natural capital. Ecol. Econom. 25 (1), 3–15.

Costanza, R., & Mageau, M. (1999). What is a healthy ecosystem? *Aquatic Ecology*, *33*(1), 105–115. doi:10.1023/A:1009930313242

Daily, Gretchen. 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*. Washington, DC: Island Press.

Dax, T. 2005. The redefinition of Europe's less favoured areas. In: Rural development in Europe – 3rd annual conference –Funding European Rural Development in 2007–2013. MPRA paper no. 711

De Aranzabal, I., Schmitz, M. F., Aguilera, P., & Pineda, F. D. (2008). Modelling of landscape changes derived from the dynamics of socio-ecological systems: A case of study in a semiarid Mediterranean landscape. *Ecological Indicators*, *8*(5), 672–685. doi:10.1016/j.ecolind.2007.11.003

Dorresteijn, I., Hartel, T., Hanspach, J., von Wehrden, H., & Fischer, J. (2013). The Conservation Value of Traditional Rural Landscapes: The Case of Woodpeckers in Transylvania, Romania. *PLoS ONE*, *8*(6), e65236. doi:10.1371/journal.pone.0065236

European commission 2008. "LIFE and Europe's grasslands: Restoring a forgotten habitat". Coordinated by the Environment Directorate-General (LIFE Unit - E.4).

European Commission, 2011. Our life insurance, our natural capital: an EU biodiversity strategy to 2020. Brussels, 2011-05-03. COM(2011) 244.

European Environment Agency (EEA), 2010. Article 17 Reports: EU CIRCA database

European Environmental Agency (EEA), 2005. Trends in birds and butterfly populations in EU-25.

European Union. 2010. Directorate-General for Agricutlure and Rural Development. Rural Development in the European Union Statistical and Economic Information. Report pp257.

FAO: T. Sakuyama: *The Roles of Agriculture, Project Brief No. 2 (Rome: 2007)*

Figueiredo J, Pereira HM (2011) Regime shifts in socio-ecological model of farmland abandonment. Landscape Ecol 26: 737–749 doi:10.1007/s10980-011-9605-3.

Findlay, A. M., Short, D., & Stockdale, A. (2000). The labour-market impact of migration to rural areas. *Applied Geography*, *20*(4), 333–348. doi:10.1016/S0143-6228(00)00012-6

Fischer, J., Hartel, T., & Kuemmerle, T. (2012). Conservation policy in traditional farming landscapes. *Conservation Letters*, *5*(3), 167–175. doi:10.1111/j.1755-263X.2012.00227.x

Fowler, P. J. 2003. UNESCO World Heritage Cultural Landscapes 1992-2002 pp140

Gadgil, M., and Berkes, F. (1991). Traditional resource management systems. Resour Manage Optimization, 8:127–41.

Gadgil, M., Berkes, F., Folke, C. (1993). Indigenous knowledge for biodiversity conservation. Ambio, 22:151–56.

Gallart, F., Llorens, P., & Latron, J. (1994). Studying the role of old agricultural terraces on runoff generation in a small Mediterranean mountainous basin. *Journal of Hydrology*, *159*(1–4), 291–303. doi:10.1016/0022-1694(94)90262-3

Gantioler, S., Rayment, M., Bassi, S., Kettunen, M., McConville, A., Landgrebe, R., Gerdes, H. & ten Brink, P. (2010) *Costs and Socio-Economic Benefits associated with the Natura 2000 Network.* Final report to the European mmission, DG Environment on Contract ENV.B.2/SER/2008/0038. Institute for European Environmental Policy / GHK / Ecologic,Brussels 2010

Garay, E. P., Larrabure, J. L. (2011). Relational knowledge systems and their impact on management of mountain ecosystems: Approaches to understanding the motivations and expectations of traditional farmers in the maintenance of biodiversity zones in the Andes. Management of Environmental Quality: An International Journal, 22 (2): 213 - 232

Gobster PH, Nassauer JI, Daniel TC, Fry G. (2007). The shared landscape: what does aesthetics have to do with ecology? Landsc Ecol 22:959–72.

Guilherme, J. L., & Miguel Pereira, H. (2013). Adaptation of Bird Communities to Farmland Abandonment in a Mountain Landscape. *PLoS ONE*, *8*(9), e73619. doi:10.1371/journal.pone.0073619

Gutman, P. (2007). Ecosystem services: Foundations for a new rural–urban compact. *Ecological Economics*, *62*(3–4), 383–387. doi:10.1016/j.ecolecon.2007.02.027

Halada, L., Evans, D., Romão, C., & Petersen, J.-E. (2011). Which habitats of European importance depend on agricultural practices? *Biodiversity and Conservation*, *20*(11), 2365–2378. doi:10.1007/s10531-011-9989-z

Harrison, P., Vandewalle, M., Sykes, M., Berry, P., Bugter, R., Bello, F., … Zobel, M. (2010). Identifying and prioritising services in European terrestrial and freshwater ecosystems. *Biodiversity and Conservation*, *19*(10), 2791–2821. doi:10.1007/s10531-010-9789-x

Haslett, J., Berry, P., Bela, G., Jongman, R. G., Pataki, G., Samways, M., & Zobel, M. (2010). Changing conservation strategies in Europe: a framework integrating ecosystem services and dynamics. *Biodiversity and Conservation*, *19*(10), 2963–2977. doi:10.1007/s10531-009-9743-y

Höchtl, F., Lehringer, S. and Konold, W. (2005). "Wilderness": what it means when it becomes a reality—a case study from the southwestern Alps. Rural Landscapes: past processes and future strategies 70(1–2), pp. 85–95. doi:10.1016/j.landurbplan.2003.10.006

Hunziker, M., Felber, P., Gehring, K., Buchecker, M., Bauer, N., & Kienast, F. (2008). Evaluation of Landscape Change by Different Social Groups. *Mountain Research and Development*, *28*(2), 140–147. doi:10.1659/mrd.0952

Kaltenborn, B. P., & Bjerke, T. (2002). Associations between environmental value orientations and landscape preferences. *Landscape and Urban Planning*, *59*(1), 1–11. doi:10.1016/S0169-2046(01)00243-2

Kasimis, Ch., Papadopoulos, A., Zacopoulou, E., (2003) Migrants in Rural Greece‟, Sociologia Ruralis, 43(2):167-184.

Keenleyside C. & Tucker, G. (2010). Farmland Abandonment in the EU: an assessment of trends and prospects. London: WWF and IEEP. 97 pp.

Kleijn, D., & Sutherland, W. J. (2003). How effective are European agri-environment schemes in conserving and promoting biodiversity? *Journal of Applied Ecology*, *40*(6), 947–969. doi:10.1111/j.1365-2664.2003.00868.x

Klimek, S., Richter gen. Kemmermann, A., Hofmann, M., & Isselstein, J. (2007). Plant species richness and composition in managed grasslands: The relative importance of field management and environmental factors. *Biological Conservation*, *134*(4), 559–570. doi:10.1016/j.biocon.2006.09.007

Körner, C. and Ohsawa, M. (eds.), 2005. Mountain systems. *In:* Hassan, R.; Scholes, R. and Ash, N. (eds.), *Ecosystems and Human Well-being: Current State and Trends, Volume 1.* Island Press, Washington DC: 681–716.

Laiolo, P., Dondero, F., Ciliento, E., & Rolando, A. (2004). Consequences of pastoral abandonment for the structure and diversity of the alpine avifauna. *Journal of Applied Ecology*, *41*(2), 294–304. doi:10.1111/j.0021-8901.2004.00893.x

Lambin, E. F., & Meyfroidt, P. (2011). Global land use change, economic globalization, and the looming land scarcity. *Proceedings of the National Academy of Sciences*, *108*(9), 3465–3472.

Larson, A. M., & Soto, F. (2008). Decentralization of Natural Resource Governance Regimes. *Annual Review of Environment and Resources*, *33*(1), 213–239. doi:10.1146/annurev.environ.33.020607.095522

Lasanta-Martínez, T., Vicente-Serrano, S. M., & Cuadrat-Prats, J. M. (2005). Mountain Mediterranean landscape evolution caused by the abandonment of traditional primary activities: a study of the Spanish Central Pyrenees. *Applied Geography*, *25*(1), 47–65. doi:10.1016/j.apgeog.2004.11.001

Lomba, A., Gerra, c., Alonso, J., Honrado, J., Jongmanb, R., McCracken, D. (2014). Mapping and monitoring High Nature Value farmlands: Challenges in European landscapes. Journal of Environmental Management, 143:140-150.

Maanen, E. van, G. Predoiu, R. Klaver, M. Soulé, M. Popa, O. Ionescu, R. Jurj, S. Negus, G.Ionescu, W. Altenburg 2006., Safeguarding the Romanian Carpathian Ecological Network.A vision for large carnivores and biodiversity in Eastern Europe. A&W ecological consultants, Veenwouden, The Netherlands. Icas Wildlife Unit, Brasov, Romania.

MacDonald, D., Crabtree, J. ., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., … Gibon, A. (2000). Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. *Journal of Environmental Management*, *59*(1), 47–69. doi:10.1006/jema.1999.0335

Machold, I., T. Dax, et al. (2002). Youth Participation in Rural Society in Maurau, Austria A Break with Traditional Patterns? T. Dax and I. Machold. Wein, Austria Byundesanstalt fur Bergbauernfragen: 103-121.

Maes, J., Hauck, J., Paracchini, M. L., Ratamäki, O., Termanse, M., Perez-Sboa, M., Kopperoinen, L., Rankinen, K., Schägner, J. P., Henrys, P., Cisowska, I., Zandersen, M., Jax, K., La Notte, A., Leikola, N., Pouta, E., Smart, S., Hasler, B., Lankia, T., Andersen, H. E., Lavalle, C., Vermaas, T., Alemu, M. H., Scholefield, P., Batista, F., Pywell, R., Hutchins, M., Blemmer, M., Fonnesbech-Wulff, A., Vanbergen, A. J., Münier, B., Baranzelli, C., Roy, D., Thieu, V., Zulian, G., Kuussaari, M., Thodsen, H., Alanen, E.L., Egoh, B., Sørensen, P. B., Braat, L. & Bidoglio, G. (2012b). *A spatial assessment of ecosystem services in Europe: methods, case studies and policy analysis. - phase 2.* PEER Report No 4. Ispra: Partnership for European Environmental Research

Maes, J., Hauck, J., Paracchini, M. L., Ratamäki, O., Hutchins, M., Termansen, M. Bidoglio, G. (2013a). Mainstreaming ecosystem services into EU policy. *Terrestrial Systems*, *5*(1), 128–134. doi:10.1016/j.cosust.2013.01.002

Maes, J., Paracchini, M. L., Zulian, G., Dunbar, M. B., & Alkemade, R. (2012). Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. *Biological Conservation*, *155*(0), 1–12. doi:10.1016/j.biocon.2012.06.016

Maes, W. H., Heuvelmans, G., & Muys, B. (2009). Assessment of Land Use Impact on

Water-Related Ecosystem Services Capturing the Integrated Terrestrial−Aquatic System. *Environmental Science & Technology*, *43*(19), 7324–7330. doi:10.1021/es900613w

Martín-López, B., Iniesta-Arandia, I., García-Llorente, M., Palomo, I., Casado-Arzuaga, I., Amo, D. G. D., … Montes, C. (2012). Uncovering Ecosystem Service Bundles through Social Preferences. *PLoS ONE*, *7*(6), e38970. doi:10.1371/journal.pone.0038970

Millennium Ecosystem Assessment (2005). Ecosystems and Human well-being: Biodiversity Synthesis. World Resources Institute, Washington, DC

Nagendra, H., Pareeth, S., Sharma, B., Schweik, C., & Adhikari, K. (2008). Forest fragmentation and regrowth in an institutional mosaic of community, government and private ownership in Nepal. *Landscape Ecology*, *23*(1), 41–54. doi:10.1007/s10980-007-9162-y

Natori, Y., & Chenoweth, R. (2008). Differences in rural landscape perceptions and preferences between farmers and naturalists. *Journal of Environmental Psychology*, *28*(3), 250–267. doi:10.1016/j.jenvp.2008.02.002

Navarro, L., & Pereira, H. (2012). Rewilding Abandoned Landscapes in Europe. *Ecosystems*, *15*(6), 900–912. doi:10.1007/s10021-012-9558-7

š, I. (2005). The impact of economic, social and political factors on the landscape structure of the Vidzeme Uplands in Latvia. *Rural Landscapes: Past Processes and Future Strategies*, *70*(1–2), 57–67. doi:10.1016/j.landurbplan.2003.10.005

Ostermann, O. P. (1998). The need for management of nature conservation sites designated under Natura 2000. *Journal of Applied Ecology*, *35*(6), 968–973. doi:10.1111/j.1365-2664.1998.tb00016.x

Paracchini ML, Petersen JE, Hoogeveen Y, Bamps C, Burfield I, van Swaay Ch (2008) High nature value farmland in Europe—an estimate of the distribution patterns on the basis of land cover and biodiversity data. Office for Official Publications of the European Communities, Luxembourg, 87 pp

Peco, B., Carmona, C. P., de Pablos, I., & Azcárate, F. M. (2012). Effects of grazing abandonment on functional and taxonomic diversity of Mediterranean grasslands. *Agriculture, Ecosystems & Environment*, *152*(0), 27–32. doi:10.1016/j.agee.2012.02.009

Pereira, E., Queiroz, C., Pereira, H., & Vicente, L. (2005). Ecosystem Services and Human Well-Being: A Participatory Study in a Mountain Community in Portugal.

Ecology and Society, *10*. Retrieved from http://hdl.handle.net/10535/3225

Pereira, H. M., & Daily, G. C. (2006). MODELING BIODIVERSITY DYNAMICS IN COUNTRYSIDE LANDSCAPES. *Ecology*, *87*(8), 1877–1885. doi:10.1890/0012-9658(2006)87[1877:MBDICL]2.0.CO;2

Plieninger, T., Höchtl, F., & Spek, T. (2006). Traditional land-use and nature conservation in European rural landscapes. *European Rural Landscapes: Land-Use, Biodiversity, and Conservation*, *9*(4), 317–321. doi:10.1016/j.envsci.2006.03.001

Portela, J. F. G., Gerry, C. (2002). From common problems to a new policy architecture: a Portuguese perspective on Mediterranean rural development. IN: B. Kasimis, B., Stathakis, G. (orgs*), The development of the countryside in Southern Europe*, Aldershot (Inglaterra), Ashgate.

Price, M. 2010. Chapter 9: Protected areas in Europe´s mountains.In:Europe´secological backbone: recognizing the true value of our mountains. EEA Report, No 6/2010.

Proença, V., & Pereira, H. M. (2013). Species–area models to assess biodiversity change in multi-habitat landscapes: The importance of species habitat affinity. *Basic and Applied Ecology*, *14*(2), 102–114. doi:10.1016/j.baae.2012.10.010

Raj Khanal, N., & Watanabe, T. (2006). Abandonment of Agricultural Land and Its Consequences. *Mountain Research and Development*, *26*(1), 32–40. doi:10.1659/0276-4741(2006)026[0032:AOALAI]2.0.CO;2

Roura-Pascual, N., Pons, P., Etienne, M. and Lambert, B., 2005. Transformation of a Rural Landscape in the Eastern Pyrenees Between 1953 and 2000. Mountain Research and Development 25(3), pp. 252–261.

Russo, D. (2006). Effects of land abandonment on animal species in Europe: conservation and management implications. Integrated assessment of vulnerable ecosystems under global change in the EU. Project report. 52 pp.

Sayadi, S., González-Roa, M. C., & Calatrava-Requena, J. (2009). Public preferences for landscape features: The case of agricultural landscape in mountainous Mediterranean areas. *Land Use Policy*, *26*(2), 334–344. doi:10.1016/j.landusepol.2008.04.003

Schils, R., Kuikman, P., Liski, J., van Oijen, M., Smith, P., Webb, J., alm, J., Somogyi, Z., van den Akker, J., Billett, M., Emmett, B., Evans, c., Lindner, M., Palosuo, T.,Bellamy, P., Alm, J., Jandl, R., Hiederer, R. (2008). Review of existing information on the interrelations between soil and climate change. Technical Report - 2008 –048.

Soini, K. and Aakkula, J., 2007. Framing the biodiversity of agricultural landscape: The essence of local conceptions and constructions. Land Use Policy 24(2), pp. 311–321.

Soliva, R., & Hunziker, M. (2009). How do biodiversity and conservation values relate to landscape preferences? A case study from the Swiss Alps. *Biodiversity and Conservation*, *18*(9), 2483–2507. doi:10.1007/s10531-009-9603-9

Stoate, C., Báldi, A., Beja, P., Boatman, N. D., Herzon, I., van Doorn, A., … Ramwell, C. (2009). Ecological impacts of early 21st century agricultural change in Europe – A review. *Journal of Environmental Management*, *91*(1), 22–46. doi:10.1016/j.jenvman.2009.07.005

Stockdale, A., Findlay, A., & Short, D. (2000). The repopulation of rural Scotland: opportunity and threat. *Journal of Rural Studies*, *16*(2), 243–257. doi:10.1016/S0743-0167(99)00045-5

Swiderska, K., 2003. Integrating local values in natural resource assessment: a review of assessment tools. IEED Natural Resources Group

TEEB (The Economics of Ecosystems and Biodiversity). 2010. The economics of ecosystems and biodiversity—ecological and economic foundations. P. Kumar, editor. Earthscan, London.

TEEB (The Economics of Ecosystems and Biodiversity). 2011. The economics of ecosystems and biodiversity in national and international policy making. P. ten Brink, editor. Earthscan, London.

TEEB (2012). The Economics of Ecosystems and Biodiversity in Local and Regional licy and Management. Edited by Heidi Wittmer and Haripriya Gundimeda Earthscan, London and Washington.

Ten Brink, P., Badura, T., Bassi, S., Daly, E., Dickie, I., Ding, H., Gantioler, S., Gerdes, H., Hart, K., Kettunen, M., Lago, M., Lan,S., Markandya, A., Mazza, L., Nunes, P.A.L.D., Pieterse, M., Rayment, M. & Tinch, R. (2011). Estimating the Overall conomic Value of the Benefits provided by the Natura 2000 Network. Final Synthesis eport to the European Commission, DG Environmental on Contract ENV.B.2/SER/2008/0038. Institute for European Environmental Policy/GHK/Ecologic, Brussels 2011

Van den Berg, A. E., & Koole, S. L. (2006). New wilderness in the Netherlands: An investigation of visual preferences for nature development landscapes.

Landscape and Urban Planning, *78*(4), 362–372. doi:10.1016/j.landurbplan.2005.11.006

Vartiainen, P. (1989). Counterurbanisation: a challenge for socio-theoretical geography. *Journal of Rural Studies*, *5*(3), 217–225. doi:10.1016/0743-0167(89)90001-6

Walker, A. J., & Ryan, R. L. (2008). Place attachment and landscape preservation in rural New England: A Maine case study. *Landscape and Urban Planning*, *86*(2), 141–152. doi:10.1016/j.landurbplan.2008.02.001

WallisDeVries, M. F., Poschlod, P., & Willems, J. H. (2002). Challenges for the conservation of calcareous grasslands in northwestern Europe: integrating the requirements of flora and fauna. *Biological Conservation*, *104*(3), 265–273. doi:10.1016/S0006-3207(01)00191-4

Watson, A., Alessa, L., & Glaspell, B. (2003). The Relationship between Traditional Ecological Knowledge, Evolving Cultures, and Wilderness Protection in the Circumpolar North. *Conservation Ecology*, *8*. Retrieved from http://hdl.handle.net/10535/7614

Westhoek, H. J., van den Berg, M., & Bakkes, J. A. (2006). Scenario development to explore the future of Europe’s rural areas. *Scenario-Based Studies of Future Land Use in Europe*, *114*(1), 7–20. doi:10.1016/j.agee.2005.11.005

Zube, E. (1997). Landscape Values. In S. Wapner, J. Demick, T. Yamamoto, & T. Takahashi (Eds.), *Handbook of Japan-United States Environment-Behavior Research* (pp. 199–208). Springer US. Retrieved from http://dx.doi.org/10.1007/978-1-4899-0286-3_14

Zube, E. H., Sell, J. L., & Taylor, J. G. (1982). Landscape perception: Research, application and theory. *Landscape Planning*, *9*(1), 1–33. doi:10.1016/0304-3924(82)90009-0

Chapter 2. Livelihoods, farmers’ perceptions and the drivers of agro-pastoral abandonment: how willing are locals to leave rural mountain communities?

Abstract

Drivers of abandonment are linked and complex, changing temporally and spatially. We present a summary of those driving forces (frictions and pressures) temporally acting locally in a northern rural mountain community, in Portugal. In addition, we determine individuals' economic thresholds to migrate outside of the community. We name this threshold the willingness to leave (WTL). We determined that geo-biophysical constrains and lack of jobs were the main pressures in the past. However, the importance of these pressures and of other pressures varied along the years. Presently, the introduction of technology and CAP subsidies is the main friction holding back abandonment and, at the same time, potentially the main trigger of land abandonment with the possible reduction of agricultural subsidies in the future. Generally, respondents indicated that they would leave for less than 1500 € a month. We found a positive correlation between willingness to leave and livestock ownership. Those individuals whose farmland utility is high are less willing to migrate. Our study provides a first of present and potential future drivers of both migration and abandonment of pastoral activities at the individual level.

Keywords: drivers, farmland utility, land abandonment, migration, socio-economic dynamics

2.1.Introduction

Since the second half of the 20th century, agricultural abandonment has become an important land use shift throughout much of Europe. In the last few decades, abandonment has become particularly important in Southwestern Europe, including Portugal, Spain and Italy (Keenlyside and Tucker 2010) This shift in land use has received much attention as it impacts positively and negatively, ecosystems and its services, as well as biodiversity (Keenleyside and Tucker 2010; MA 2005; Navarro and Pereira 2012; ReyBenayas et al. 2007; Schrötter et al. 2005). Shifts in land-use is also believe to have profound implications for food security becoming important aspects of rural sustainable development. Land use shifts are a result of a combination of ecological and socio-economic driving forces (Rey Benayas et al. 2007). Determining the spatial patterns and those drivers of land abandonment is crucial to improve land use planning, the environment and potentially the economy.

In Europe, most abandoned lands are linked to unfavourable environmental conditions (e.g. higher elevation, poor soils and steeper slopes), mainly remote mountain areas consisting of small farms (Baldock et al. 1996; MacDonald et al. 2000). Identifying the drivers of land use change requires an understanding of what causes people to make these decisions. Most of the knowledge obtained along the years has been focused primarily on macro-scale driving forces (Prishchepov et al. 2013;Baumann et al. 2011;Rey Benayas et al. 2007) and thus, we are still lacking studies carried at the individual level.

Portugal is a prime example of a country which has endured on-going abandonment since the 1950´s, specifically in rural mountain areas. This trend is a consequence of an intermingling of drivers, starting with the implementation of agrarian policies such as the " Wheat Campaign" in the 1930´s, that centered their attention on the control of prices and direct aid to areas of large wheat production, neglecting rural mountain communities that cultivated corn and rye (Aguiar et al. 2009). Starting around the same period and lasting until the mid 1960's, the population had to face reduced land for pasture after the appropriation of the land by the state, to promote afforestation through the implementation of the "Plano de Povoamento Florestal". These and many other restrictions triggered farmland abandonment and the migration of residents was accentuated as jobs in the city payed better and increased mechanization required less labour (MacDonald et al. 2000).

Drivers can be classified as locally specific, national and globally driven, varying spatially and temporally (Bürgi et al. 2004). Local drivers include those related to bio-physical constrains of the landscape, such as climate conditions, elevation and accessibility. At the national scale, these drivers become more socio-economic, and globally, drivers are mainly market oriented. In Europe, rural areas make up 57% of the territory and one third of this land is considered economically marginal, due to small farm structures and lack of market acess (Baldock et al. 1996). Since the 1960´s, rural population has decreased by 17% (FAOSTAT 2010) and the over representation of older age classes in the rural population has increased, with little succession of generations, amplifying agro-pastoral abandonment (Bernúes et al. 2005). Still, in 2011, 24% of the population resided in rural areas (Eurostat 2011).

This depopulation trend has not gone unnoticed in political discussion, ans had lead to the implementation of multiple measures in the Common Agricultural Policy (CAP), particulalry the Less Favored Areas (LFAs-Regulation 1257/1999) and agri-environmental payments, but they have been largely unsuccessful at halting this tendency. In fact, recent model projections reveal an increase throughout Europe over the next 30 years with lowered levels of CAP support (Keenleyside and Tucker 2010).

Presently, less favored areas make up half of the utilized agricultural areas (MacDonald et al. 2000) in Europe, and are characterized by its biophysical limitations.

The migration from rural to urban areas has become most pronounced in regions which are marginally economic. According to the International Migration Organization, migration is defined as a process of moving, either across an international border, or within a state. It is a population movement, encompassing any kind of movement of people, whatever its length, composition and cause (Perruchoud 2004). Rural exodus has become a primary social driving force behind land use change. A recent mathematical model has linked socio-ecological dynamics to explore farmland abandonment as a collective behavior (Figueiredo and Pereira, 2011). In this model, migration is viewed as a collective behavior that is socially and economically driven. In this model individuals decision to migrate is based on a threshold of the number of neighbours which have migrated and the threshold is dependent on the individuals utility of the agricultural area versus the utility of living in the city. In the present study, we will be using this model framework of the economics and social bonding of migration, to analyse our data. Nonetheless, to fully understand what drives an individual to abandon agricultural activities and migrate; we need to identify each individual’s threshold to various determinants. The Neoclassical Economic Theory was the first model to explain migratory movements (Arango 2000). The model states that an individual’s decision to migrate is to improve an economic situation. The main purpose of our research was to explore the perception of underlying, past, present and future determinants acting as pressures, frictions and triggers of agro-pastoral abandonment. Additionally, to determine the decision process of abandonment and migration, we analyze each individual’s threshold based on the economic utility of farmland, using Figueiredo and Pereira (2011) model. We hypothesize that those individuals who are able to maintain agricultural utility high are less willing to migrate.

2.2 Methods

2.2.1. Study area

The Castro Laboreiro parish is situated in the northeastern part of the Peneda-Gerês National Park (42ºN and 8º10’W) in Portugal, covering a total area of 9440 ha (Figure 2.1). Altitude ranges from 300 m to 1340 m with daily average temperatures of 0-2ºC in the winter and 25-28ºC in the summer. This region was unique in Portugal due to the annual transhumance. Castro Laboreiro has over 40 population settlements, organized into three different classes of occupation: 20 summer villages (brandas), 21 winter villages (inverneiras) and 8 fixed villages. The summer villages or "brandas" are located at higher elevations, in the plateau, with abundant pasture land. The inverneiras are located at lower altitudes, deepi in the valleys. The vertical movement of livestock and families from plateau settlements to valley areas occurred in December as weather conditions in the plateau became very unfavorable for agricultural activity. In the valley the families found shelter (eg.g less days of snow) and some pastures for the livestock. This seasonal subsistence strategy allowed locals to exploit two distinct ecological realities creating a heterogeneous landscape. Today, this annual transhumance has mostly stopped, driven by social-economic changes, with most of the permanent housing occurring in the plateaus and most of the valley villages becoming abandoned.

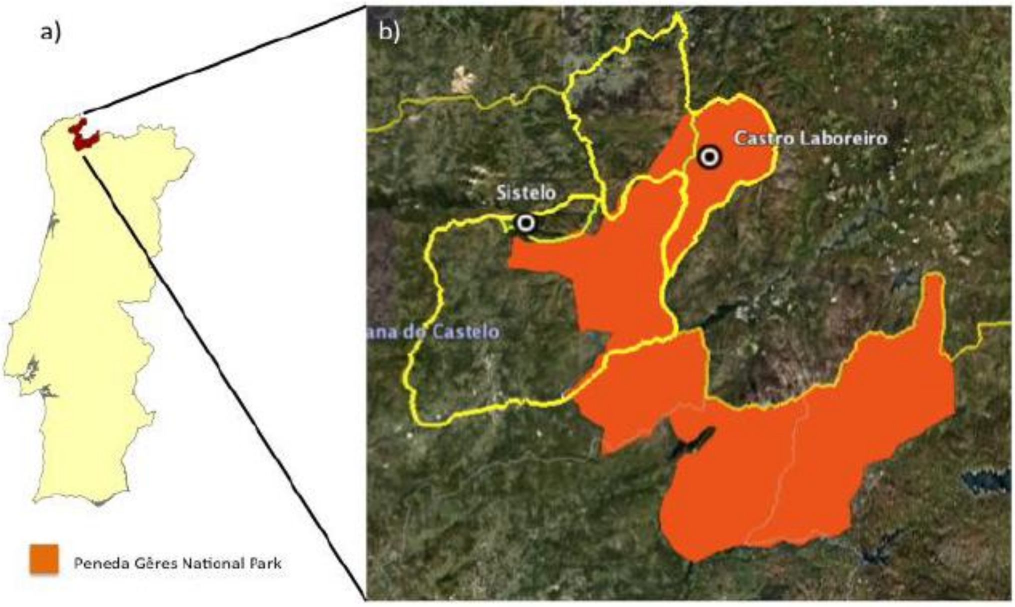

Figure 2.1 a) Location of the Peneda-Gerês National Park in Portugal b) The parish of Castro Laboreiro.

Historically subsistence based agro-pastoralism was the main activity in the region. The main crops were potatoe and rye. Farmland is divided in small parcels throughout the territory. Families hold an average of 10 to 15 parcels and the average parcel size is < 2ha (Geraldes 1996). Up until the 1960´s the economy of the Castro Laboreiro social-ecological system was largely dependent on local provisioning services.

Although agro-pastoral activities are still apparent throughout the region, the dynamics and the number of people who still have livestock and cultivate has decreased. In the late 50´s each household had on average a minimum of 20 to a maximum of 80 heads of livestock, where today the average household has <2 farm animals (Geraldes 1996). Between 1989 and 2009, bovine, goats and sheep have decreased by 46%, 61% and 51%, respectively (INE 2009). Population numbers have also decreased by 37% between 1960 and 2011 (INE 1960 and 2011) with a present population of 540 inhabitants, mainly represented by women and elderly (Figure 2.2). These changes to the socio-ecological systems have impacted the landscape, where today, scrubland has become the most dominant land cover and agricultural areas has decreased by 51.5% since the late 1960´s (Rodrigues 2010). Most of the abandonment occurred from 1960 to 1990.

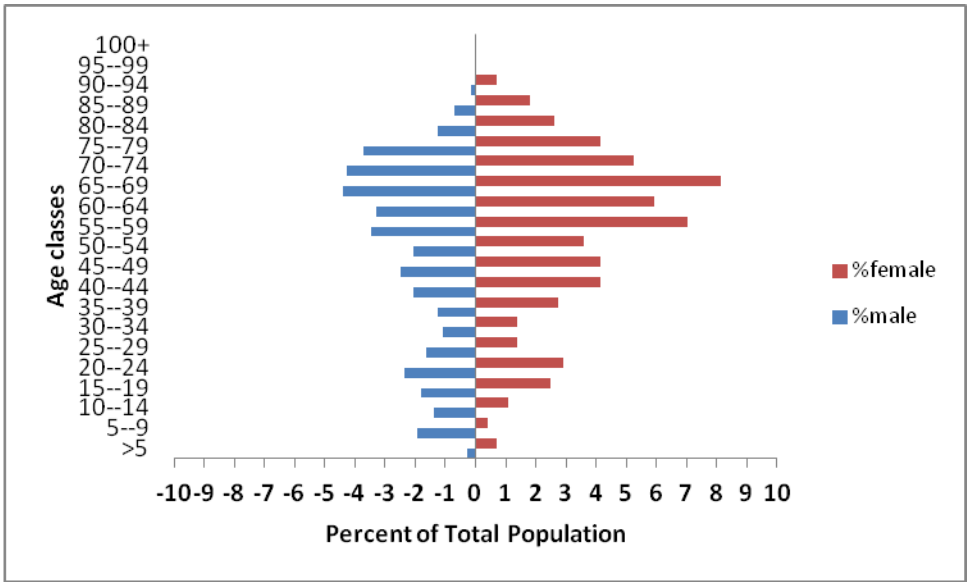

Figure 2.2 Age-structured pyramid of the Castro Laboreiro population. Source: INE 2011.

Although the agro-pastoral activity has decreased, tourism activity has steadily increased in the last two decades. The region's natural capital of a diversified landscape and an unique cultural history is an important attraction. Part time residents and weekend homeowners have increased. Presently, we can find several lodges in Castro Laboreiro along with restaurants and small businesses, mainly focusing on the tourism industry. However, oublic services have decreased along the years. Today, there are no functioning schools or healthcare centers; the closest are located approximately 30 km away in Melgaço. Although, there is an interest in developing new strategies for economic development in the region, there still remains much disagreement among the various stakeholders. The economic, environmental and social issues represent a reality found throughout much of Europe´s rural mountains, making Castro Laboreiro a representative example. In the last 50 years Castro Laboreiro´s landscape has undergone several changes.

2.2.2. Survey and data analysis

A total of 27 surveys were carried out during the months of August of 2010 and February of 2011. All locals interviewed were at present or in the past leisure or full time farmers and included both males and females over the age of 20. Informants were previously selected by a chief informant previously briefed of the studies objectives. The interviews were carried out individually at the interviewees' residence and/or in the field. During the interviews a basic set of questions (Table 2.1) was often expanded as additional ideas and concerns surfaced. On average the duration of each interview was over an hour.

Table. 2.1 Base questions explored in the individual interview.

1.) Why have you maintained (or stopped) agriculture activity?
2.) Will you continue farming in the future, and why?
3.) Why did some residents stopped agricultural activity and why do you think others continue farming?
4.) Would you consider leaving Castro Laboreiro? Why?
5.) How much money would you need to receive to leave Castro Laboreiro? What other incentives would drive you to leave Castro Laboreiro?
6.) Would you leave Castro Laboreiro if your neighbours were to migrate?

The interview questions were structured according to the studies main objectives (Table 2.1). In the first, second, and fourth question we wanted to determine the past, present and future drivers holding back farmland abandonment and those leading to abandonment at the individual level. As well as those drivers leading to migration. Based on the answers obtained from the interviews we classified the drivers into 3 categories: pressures, frictions and attractors. These categories were determined using Eiter and Potthoff (2007) and Bürgi et al. (2004) classification. In the present analysis, pressures are defined as those factors pushing for change (Eiter and Potthoff 2007), straining current land use and therefore contributing to farmland abandonment. These pressures were then sub-divided into the following categories: economic, political, cultural and technical. Frictions are those determinants that slow down land use change holding back abandonment (e.g. cultural heritage). We also identified those determinants that could be considered attractors, which are specific place conditions and triggers which can either be visible or inivisible, originating land use change (Bürgi et al. 2004).

Although, we did not make reference to a specific year, residents acknowledged that it was during the 1960´s that local rural exodus occurred. We used that year as a baseline. In the last two questions, we analyzed each individual's threshold. First we determined the economic threshold and the social connection of each individual to their neighbours. Note that the economic value given by each individual can either represent a job offer or a pension value per month for those who are no longer active in the farming population). This value is described in this study as local's willingness to leave (WTL). The provided monetary value given by each invidivual was used as a proxy representing local's farmland utility (i.e. higher values represent those farmers who are less willing to leave). In the last question, individuals were asked to provide a yes or no answer with an explanation regarding their responses.

Descriptive statistics and frequencies were used for a first analysis of the data. Based on the individual interviews, we first determined those influential drivers in the past, present and future. The pressures, frictions and attractors were categorized and summed for all interviews, giving as total for each driver in the past, present and future. The analysis then focused on exploring the relationship between present and future determinants influencing local's willingness to leave (WTL). We also explore the possibility and the presence of existing correlations between drivers. Our groups of variables consisted of mainly socio-economic and demographic characteristic identified by each respondent. We tested the relationship between the dependent variable (WTL= willingness to leave) and the various predictors using regression models. All models were calculated using the R statistics software, Version 2.0.1 (R Development Core Team, 2005).

2.3. Results

2.3.1. Pressures vs. frictions through time and attractors

Based on the interviews, we summed the most influential pressures and frictions in Castro Laboreiro and ranked them by order of importance (Fig. 2.3). Generally, farmers age and agro-pastoral activities no longer profitable were identified as the main incentives to abandon. On the other hand necessity, subsidies and sentimental value were identified as incentives to continue the activity. The pressures and frictions outlined in Fig. 2.3 provide an overview of the most influential drivers, however, for this first analysis we did not consider temporal change. In order to understand how these pressures and frictions fluctuate through time, we categorized and determined the most influential past, present and future pressures and frictions. To evaluate the trends on the various frictions and pressures through time, we summed the data obtained from the base questions from each individual (Figure 2.4). Based on the data and perception given by each individual, our results highlight alterations of both frictions and pressures through time. The most influential frictions in the past include level of education, necessity for subsistence and community work. Participants mentioned that in the past the majority of the families did not have the resources to maintain their children in school and as a result, the involvement of the children in the family's daily agro-pastoral duties was fundamental. Additionally, the social cohesion of neighbors was not only culturally relevant but also permitted extra help in the year round agro-pastoral activities. Presently and in the future, the main frictions were identified as the introduction of technology, such as mechanical equipment (e.g. tractor) which help in the daily tasks, but has lead to a decrease in labor force. Several individuals stated that "although tractors have made agricultural practices much easier it has consequently led to a decrease in the community's social-cohesion". Additionally, our results highlight that the granting of CAP subsidies is and will continue to be a fundamental friction in the continuation of agro-pastoral activities. One local stated (interview #12) "*If they cut my subsidies, I will sell all my animals and go live with my daughter. My pension is not enough to cover all my medical expenses. I am not the only one in this situation, my neighbours are the same"*. Another individual also quoted (inverview #7) *"I am young. I am only 25 years old, if they cut my subsidies what choice do I have, but to leave and look for a job. I love my job, but I have to feed my family".* Even so, the subsidies cuts are most influential for younger farmers as opposed to older farmers, whose main frictions remain the historical and cultural connection to the land.

Geo-biophysical constraints and lack of jobs were the main pressures identified in the past. Presently, these pressures remain influential; however, the number of pressures has increased. Farmers' age, lack of community work or participation (i.e. the people living in one locality no longer aiding their neighbors in agricultural activities), lack of goods and services, are additional pressures which are thought to remain important in the future. Furthermore, residents identified tourism (potential job opportunities in tourism leading to a decrease in agro-pastoral activities), destruction of crops by wild boar and horses as pressures which can contribute to agro-pastoral abandonment.

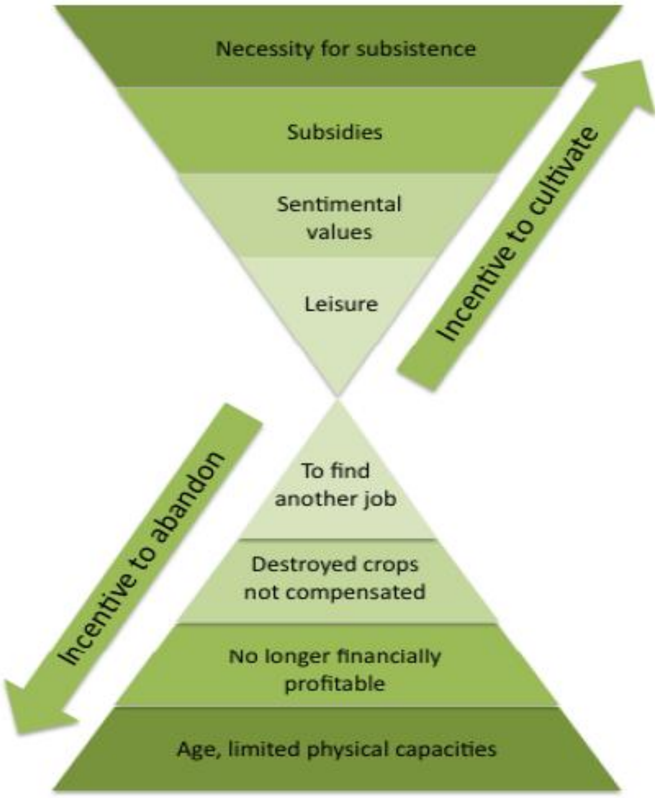

Figure 2.3. Simplified ranking of the different pressures and frictions promoting and halting agro-pastoral abandonment in Castro Laboreiro.

Presently there are more pressures contributing to agro-pastoral abandonment and fewer frictions holding back abandonment (Figure 2.4). Our findings also unveiled that certain pressures and frictions have a tipping point at which time a pressure becomes a friction and vice versa. For instance farmer's age can be both a pressure and a friction. As many individuals stated (interview # 6) "*When you are young, you are able to work the land and physical limitations are not an obstacle. However, as you get older* *you know longer have the strength to work all the land or work at all and we have no choice but to abandon agricultural activities".* Individuals also stated (direct quote from interview #23), *"In the past (1960) it was the young and married men that migrated out of the community in search of better job opportunities and the older family members, wives and children stayed behind*". Thus, in the past, age was associated to abandonment and also related to an individual's physical capacity to maintain agricultural activity. Where, today the older an individual gets the greater the pressure to abandon land.

Overall, we identified the physical conditions for agro-pastoral activities as the main past and future attractor of abandonment. This specific attractor was not found to vary through time as its impact on land abandonment remains temporally constant. Historically, the lack of available jobs leading to high rates of migration out of rural areas has been identified as the main trigger which has remained a constant pressure in Castro Laboreiro. It is relevant to point out that triggers that persist through time can be classified as a pressure or a friction (Beilin et al. 2014). In fact, from early 1960´s to 2011, Castro Laboreiro suffered a 55% in population numbers (INE 2011).

2.3.2. Economic utility: Willingness to leave (WTL)

Overall, 55.6 % of the respondents indicated that they would leave for less than 1500€ and 29.6% between 1500€ and 2500 € per month. We found a relationship between willingness to leave and the following predictor variables: age, gender, job, residency, livestock number, number of inviduals dependent on agricultural activity. The regression model revealed that the vast majority of the data variance can be explained by the various predictors (R^2=0.958). ANOVA results revealed a high significance level (F=51.676; p=0.001). We summarize the results of the various predictors, identifying the most relevant in terms of explaining locals' willingness to leave (Table 2.2). Beta values reveal that the number of livestock is the most important and also the most statistically significant coefficient (t=13.171; p=0.001).

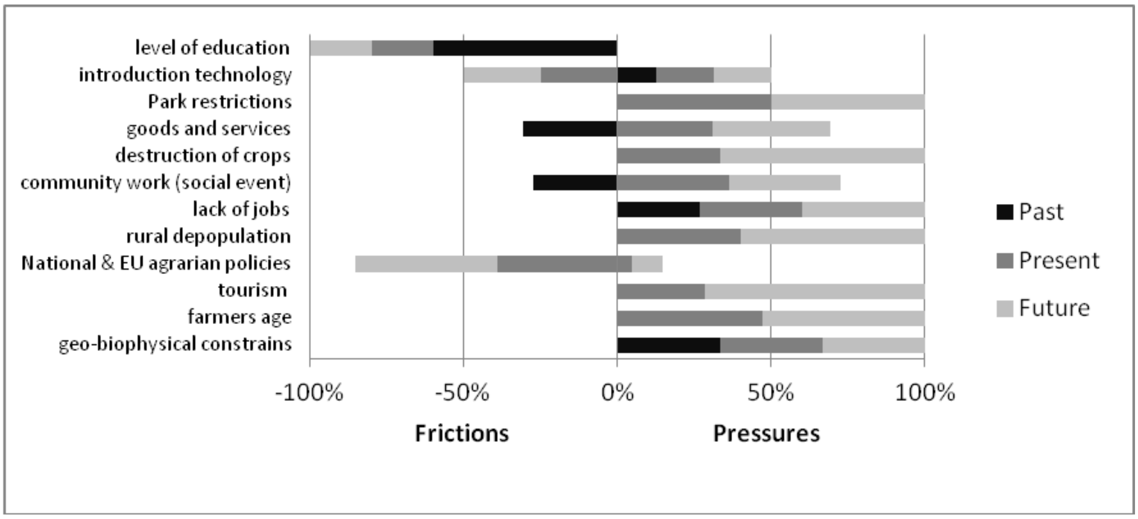

Figure 2.4. Temporal fluctuations of pressures and frictions of agro-pastoral abandonment, perceived by local inhabitants.

Table 2.2 Multiple regression results for the various predictors.

Model	Unstandardized Coefficients		Standardized Coefficients	t	Sig.
	B	Std. Error	Beta		
(Constant)	99.348	488.447		.203	.841
civiil-parish (residency)	50.635	31.335	.092	1.616	.124
Age	5.239	8.383	.048	.625	.540
Gender	-209.099	251.272	-.067	-.832	.416
Profession	-86.641	116.989	-.044	-.741	.468
duration of residency	10.501	6.289	.138	1.670	.112
individuals family	-50.248	71.587	-.040	-.702	.492
number of people dependent on agriculture	-790.785	189.921	-.388	-4.164	.001
number of livestock	93.965	7.134	1.323	13.171	.000

a. Dependent Variable: willingness to leave

There is a strong positive correlation between the number of livestock on an individual's willingness to leave (Fig. 2.5, R^2=0.890, p=0.001). In general, those rearing large number of livestock (>40) are less willing to leave than those with low levels of livestock (10). This suggests that those individuals, whose farmland utility is high due to subsidies, are less willing to migrate.

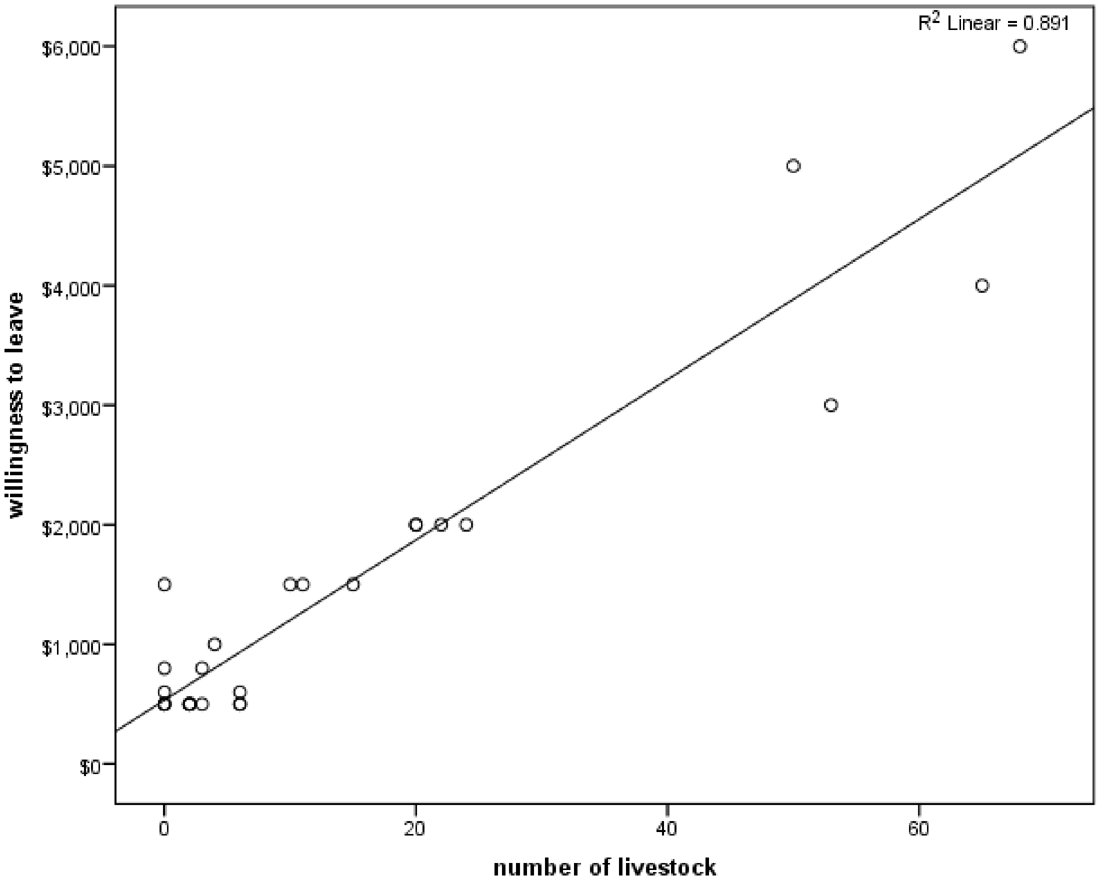

Figure 2. 5 Scatterplot of willingness to leave and livestock numbers.

We further analyzed the existence of a relationship between the various predictors. An association was found between those individuals receiving CAP subsidies and the type of agricultural practice (X^2=14.760; p=0.022). Farmers receiving subsidies, carry out more modern type agriculture, than those receiving lower subsidies. Our findings also revealed a relationship between number of livestock and type of agriculture (X^2=44.325, p=0.026). Those individuals with low livestock numbers practice a more traditional agro-pastoral activity.

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Those individuals with low livestock numbers practice a more traditional agro-pastoral activity.

In the last question of the interview, individuals were asked if they would consider migrating if their neighbors left. Our results show that, 89% would not migrate if indeed their neighbours decided to migrate. According to interview #17 "*We are not dependent on what our neighbours do, each person does what they want*". However, we also encountered situations of individuals who had already moved residency from summer villages to fixed villages, mainly due to the out migration of surrounding neighbours. "*I had to leave, everyone around me left, I was alone and nobody to talk to, if something happened to me nobody would know*" (interview #9). In addition, 96% of the respondents stated that the main driver behind the discontinuation of the traditional transhumance was related to the decrease in the number of individuals actually residing in the winter villages. Of those interviewed only 1 family continued the tradition. The head of the family stated "*It is part of our tradition as long as I am physically able, I will continue. We only have six cows now but, the animals prefer it there in the winter, and so do we*" (interview #20).

2.4. Discussion

2.4.1. Would currently active farmers stop agro-pastoral activities?

Our research offers new insights on how pressures and frictions have fluctuated and evolved through time influencing individuals' decisions to maintain or stop agro-pastoral activities and migrate. Our results show that the primary past causes of rural urban migration was driven by economic opportunities, a trend, similar to other European rural mountain communities (Mottet et al. 2006, Gellrich and Zimmermann 2007). Today, this pressure remains influential; however, the number of pressures has increased along the years. Other studies have shown that those who are more likely to migrate are generally the younger and better educated (Simon et al., 2010). As a result, rural population has decreased and aged with lack of intergenerational succession contributing to agro-pastoral abandonment (Bernués et al., 2011). This trend is expected to continue in the EU-27, as recent data show that 55% of farm holders are 60 years of age and older, raising to 73.4% in Portugal (EUROSTAT, 2007). In addition, limited social amenities (schools and health centers) have not only contributed to the out migration of rural residents, but have also difficulted past and present re-establishments (Pereira et al., 2005).

In the past, cultural value, community work and level of education (i.e low secondary education) were key frictions contributing to the preservation of Castro´s social-ecological system. However, the importance of these resistance factors has decreased, due to the outmigration of younger generations, resulting in a decline in the agricultural labour force. Presently, CAP subsidies have been identified as the primary friction holding back abandonment, suggesting that the decline of agro-pastoral activities is mitigated to some extent by CAP subsidies. Therefore, a reduction in subsidy payments is a possible future trigger of pastoral abandonment (Keenleyside and Tucker 2010). Presently, there are more pressures to abandon agricultural activity, outweighting the frictions. Although, not analyzed in the current study, competition from global markets is thought to increase the risk of abandonment in low intensity livestock systems, however, we are presently unable to determine the impact, if any, in the present study area.

2.4.2. Rural utility vs. city utility: primary determinants of local's willingness to leave

Overall, we found a strong negative correlation between local's willingness to leave and agricultural utility. Our findings suggest that local's willingness to leave has an economic threshold, which in our study area is based on subsidy payments for livestock rearing. According to Figueiredo and Pereira (2012) model on the socioeconomics of migration, when farmland utility is high, individuals are less likely to migrate for economic reasons. In the model, social-bonds provide further friction and make migration of an individual less likely unless a large part of the community decides to migrate. Our data supports part of this assumption and revealed that individuals with more than 50 heads of livestock would only consider migrating and abandoning the activity for € 3000 or more per month and those with a lower number of livestock would leave for less. Our study also suggests that one of the primary determinants behind the discontinuation of the seasonal migration and of the abandonment of a number of winter villages was motivated by social-bonds.

In our study we were not able to determine if past agro-pastoral abandonment resulted in the out migration of an individual. Determining this would be difficult as we did not carry out interviews with those who have already migrated, however, we did verify that presently, younger residents would migrate if CAP subsidies were reduced. However, a Swiss study divulged that land abandonment has no association to migration (Gellrich and Zimmermann 2007). These findings may not be common to all of Europe and are in fact controversial, as statistical data show otherwise.

2.4.3. Future trends and the benefits of agro-pastoral activities

According to recent modelling studies, farmland abandonment in Europe is estimated to increase between 0.7 % (2020 Regionalisation Scenario) to 6.7% (EURURALIS Global Cooperation Scenario) over the next 30 years (Keenlyside and Tucker 2010). In addition, 28.1% of abandoned land is thought to be grassland of high nature value (HNV) (Paracchini et al., 2008). This is a major concern in Portugal, as 57.6% of farmland is categorized has HNV (Paracchini et al., 2008). Overall, countries such as Finland, Sweeden, Pyrenees, north-western Spain and Portugal, the Massif Central, Apennines and the Alps have been identified as areas of high risk from abandonment.

According to our calculations and based on the individuals questioned in the present study, on average each family has less than 2 farm animals per family. In the late 50´s each household had a minimum of 20 to 80 heads of livestock (Geraldes 1996). This trend has become evident throughout much of Europe, where between 1990 to 2010, livestock numbers, decreased by 25% (FAOSTAT, 2010). Studies have shown decreases in grazing pressure will lead to the natural successional evolution of vegetation, decreasing habitat diversity and opportunity for specific wildlife (Laiolo et al., 2004, Galop et al., 2011, Rescia et al., 2008), as well as increasing the potential for fire (Moreira et al., 2001). Furthermore, our study highlights that farmers with higher livestock numbers have adapted a more modern agricultural practice, which could potentially jeopardizing the sustainaibiliy of specific biodiversity dependent on a more traditional agricultural practice (Ostermann 1998).

There are several challenges policy makers and local management planner's face today and in the future. Understanding the causes or factors of landscape change has become important to know which driving forces can be controlled by different planning authorities. However, this is not always easy as our study indicates that driving forces fluctuate through. The implementation of CAP subsidies and the introduction of LFA (Less Favored Areas) has had little impact on the maintenance of low-intensity systems in marginal rural mountains. Nonetheless, in Castro Laboreiro, there are a number of farmers who have taken advantage of the situation and have maintained economically viable by increasing their number of livestock. Additionally, over the last two decades, Castro Laboreiro has seen an increase in tourism activity, which has resulted, although not always the case, in land abandonment by those who found jobs in the sector. Throughout European mountain communities, traditional agricultural landscapes are positively influenced through rural tourism (Vanslembrouck et al. 2005) and are an adaptive alternative, promoting rural development and contributing to the local economy (Rescia et al. 2008). Yet, if there is a constant increase in land abandonment, rural tourism could potentially decrease if the main attraction are traditional landscapes or will require a new angle focusing on valuing a more natural setting promoting wilderness. In general, future decision making, policy and planning need further insight on how various driving forces, such as policital and economic instruments as well as bio-physical constraints play a role in halting and promoting abandonment.

2.5. References

Aguiar, C., Rodrigues, O., Azevedo, J., Domingos, T. (2009) Montanha, pp 295-339 *in* Pereira, H. M., Domingos, T., Vicente, L., Proença, V., Eds.), Ecossistemas e Bem-star Humano: Avaliação para Portugal do Millenium Ecosystem Assessment (Pereira, H. M., Domingos, T., Vicente, L., Proença, V., Eds.), Escolar Editora.

Anthrop, M. 2005. Why landscapes of the past are important for the future. Landscape Urban Planning. 70:21-34.

Arango, J. 2000. “Explaining Migration: A Critical View”, International Social Science Journal, 52: 283–296

Baldock, D., Beaufoy, G., Brouwer, F., Godeschalk, F., 1996. Farming at the Margins:Abandonment or Redeployment of Agricultural Land in Europe. Institute for European and Environmental Policy and Agricultural Economics Research Institute, London

Baumann M et al 2011 Patterns and drivers of post-socialist farmland abandonment in Western Ukraine Land Use Policy, 28:552–62

Bell, S., Montarzino, A., Aspinall, P., Penéze, Z., Nikodemus, O. 2009. Rural society, social inclusion and landscape change in central and eastern Europe: A case study of Latvia. Sociologia Ruralis 49(3):295-326.

Bernués A., Riedel J.L., Asensio M.A., Blanco M., Sanz A., Revilla R. and Casasús I., 2005. An integrated approach to studying the role of grazing livestock systems in the conservation of rangelands in a protected natural park (Sierra de Guara, Spain). In: Livestock Production Science, 96. p. 75-85

Bernués, A., Ruiz, R., Olaizola, A., Villalba, D., Casasús, I. 2011. Sustainability of pasture-based livestock farming systems in the European Mediterranean context: Synergies and trade-offs. Livestock Science, 139:44-57.

Bürgi, M., Hersperger, A., & Schneeberger, N. (2004). Driving forces of landscape change — current and new directions. *Landscape Ecology*, *19*(8), 857–868. doi:10.1007/s10980-004-0245-8

Campbell, D. J., Lusch, D. P., Smucker, T. A., Wangu, E. E. 2005. Multiple methods in the study of driving forces of land use and land cover change: a case study of SE Kajiado district Kenya. Human Ecology 33, 763-794.

Casasús, I., Bernués, A., Sanz, A., Villalba, D., Riedel, J. L., Revilla, R. 2007. Vegetation dynamics in the Mediterranean forest pastures as affected by beef cattle grazing. Agr. Ecosystem Environment, 121:365-370.

EEA 2004. High Nature Value Farmland: characteristics, trends and policy challenges. European Environmental Agency, Copenhagen, p. 27.

EUROSTAT.2007. Farm Structure Survey.

EUROSTAT 2011. Farm Structure Survey

FAOSTAT. 2010. Livestock

FAOSTAT. (2011). http://faostat.fao.org.

Eurostat (2011a). Population, Date of Access 20.07.11, Available from http://epp.eurostat.ec.europa.eu/portal/page/portal/population

Figueiredo J, Pereira HM (2011) Regime shifts in socio-ecological model of farmland abandonment. Landscape Ecol 26: 737–749 doi:10.1007/s10980-011-9605-3.

Figueiredo, J.M. (2005), Fluxos Migrat/orios e Cooperação para o Desenvolvimento: Realidades compat/ıveis no contexto Europeu?, Ph.D. Thesis, Universidade Tecnica de Lisboa, Lisbon. Online version (accessed 27 July 2010): http://www.oi.acidi.gov.pt/modules.php?name=News&file=article&sid=856.

Galop, D., Houet, T., Mazier, F., Leroux, G., Rius, D. 2011. Grazing activities and biodiversity history in the Pyrenees: New insights on high altitude ecosystems in the framework of a Human-Environment Observatory. Pages *news*: vol.12 (2).

Gellrich, M., & Zimmermann, N. E. (2007). Investigating the regional-scale pattern of agricultural land abandonment in the Swiss mountains: A spatial statistical modelling approach. *Landscape and Urban Planning*, *79*(1), 65–76. doi:10.1016/j.landurbplan.2006.03.004

Geraldes, A. D. 1996. Brandas e Inverneiras. Particulariedades do sistema agro-astoril crastejo. Cadernos Juríz Xurés. Instituto da Conservação da Natureza e Parque Nacional Peneda-Gerês.

Halada, L., Evans, D., Romão, C., Petersen, J. 2011. Which habitats of European importance depend on agricultural practices? Biodiversity Conservation 20:2365-2378.

Hatna, E., & Bakker, M. (2011). Abandonment and Expansion of Arable Land in Europe. *Ecosystems*, *14*(5), 720–731. doi:10.1007/s10021-011-9441-y

JNCC. 2007. Second Report by the UK under Article 17 on the implementation of the Habitats Directive from January 2001 to December 2006. JNCC, Peterborough. Available from www.jncc.gov.uk/article17

Keenleyside C. & Tucker, G. (2010). Farmland Abandonment in the EU: an assessment of trends and prospects. London: WWF and IEEP. 97 pp.

Laiolo, P., Dondero, F., Ciliento, E., & Rolando, A. (2004). Consequences of pastoral abandonment for the structure and diversity of the alpine avifauna. *Journal of Applied Ecology*, *41*(2), 294–304. doi:10.1111/j.0021-8901.2004.00893.x

Lasanta-Martínez, T., Vicente-Serrano, S. M., Cuadrat-Prats, J. M. 2005. Mountain Mediterranean landscape evolution caused by the abandonment of traditional primary activities: a study of the Spanish Central Pyrenees. Applied Geography, 25:47-65.

MacDonald, D., Crabtree, J. ., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., … Gibon, A. (2000). Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. *Journal of Environmental Management*, *59*(1), 47–69. doi:10.1006/jema.1999.0335

Mertz, O., Wadley, R. L., Christensen, A. E. 2005. Local land use strategies in a globalizing world: Subsistence farming, cash crops and income diversification. Agricultural Systems 85:209-215.

Millinnium Ecosystem Assessment (MA) 2005 Ecosystems and human well-being: Synthesis. Washington, DC.: Island Press, 137p.

Moreira, F., Rego, F., & Ferreira, P. (2001). Temporal (1958–1995) pattern of change in a cultural landscape of northwestern Portugal: implications for fire occurrence. *Landscape Ecology*, *16*(6), 557–567. doi:10.1023/A:1013130528470

Mottet, A., Ladet, S., Coqué, N., Gibon, A. 2006. Agricultural land-use change and its drivers in mountain landscapes: A case study in the Pyrenees. Agriculture, Ecosystems and Environment, 114:296-310.

Mouillot, F., Ratte, J.P., Joffre, R., Moreno, J.M., Rambal, S., 2003. Some determinants of the spatiotemporal fire cycle in a mediterranean landscape (Corsica, France). Landsc. Ecol. 18, 665–674.

Naylor, R., Ehrlich, P. 1997. Natural pest control services and agriculture. In: Zhang, W., Ricketts, T. H., Kremen, C., Carney, K., Swinton, S. M. 2007. Ecosystem services and dis-services to agriculture. Ecological Economics, 64:253-260.

Nunes, A. N., Coelho, C. O. A., de Almeida, A. C., & Figueiredo, A. (2010). Soil erosion and hydrological response to land abandonment in a central inland area of Portugal. *Land Degradation & Development*, *21*(3), 260–273. doi:10.1002/ldr.973

Ostermann, O. P. 1998. Forum: Biodiversity & High Nature Value Farmland. The need for management of nature conservation sites designated under Natura 2000. Journal of Applied Ecology, 35:968-973.

Paúl, C., Fonseca, A. M., Martín, I., & Amado, J. (2003). Psychosocial Profile of Rural and Urban Elders in Portugal. *European Psychologist*, *8*(3), 160–167. doi:10.1027//1016-9040.8.3.160

Paracchini, M. L., J.-E., Petersen, Y., Hoogeveen, C., Bamps, I., Burfield and C. van Swaay (2008), High Nature Value Farmland in Europe. An Estimate of the Distribution Patterns on the Basis of Land Cover and Biodiversity Data. European Commission Joint Research Centre, Institute for Environment and Sustainability. Report EUR 23480 EN. 87 p. Office for Official Publications of the European Communities, Luxembourg.

Peixoto, J. (2004), "As Teorias Explicativas das Migrações: Teorias Micro e Macro-ciol/ogicas",SOCIUS Working Papers, 11/2004, Lisbon (SOCIUS). URL (accessed 27 July 2010) :http://pascal.iseg.utl.pt/~socius/publicacoes/wp/wp200411.pdf.

Pereira, E., Queiroz, C., Pereira, H. M., & Vicente, L. (2005). Ecosystem Services and Human Well-Being: a Participatory Study in a Mountain Community in Portugal.

Ecology and Society, *10*(2). Retrieved from http://www.ecologyandsociety.org/vol10/iss2/art14/

Perruchoud, R., ed. (2004), Glossary on Migration, International Migration Law, 1, Geneva (International Organization for Migration).

PRODER, 2007. Novas Medidas: Agro-ambientais e Silvo-ambientais, Informações técnicas: Intervenções territoriais integradas (Peneda-Gerês).

Rescia, A. J., Pons, A., Lomba, I., Esteban, C., & Dover, J. W. (2008). Reformulating the social–ecological system in a cultural rural mountain landscape in the Picos de Europa region (northern Spain). *Landscape and Urban Planning*, *88*(1), 23–33. doi:10.1016/j.landurbplan.2008.08.001

ReyBenayas, J. M., Martins, A., Nicolau, J. M. and Shulz, J. J. 2007. Abandonment of agricultural land: an overview of drivers and consequences. CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources, 2(57):1-14.

Ruiz-Mirazo, J., Robles, A.B., Gonzáez-Rebollar, J.L., 2009. Pastoralism in natural parks of Andalusia (Spain): a tool for fire prevention and the naturalization of ecosystems. Options Méditerr. Sér. A 91, 141–144.

Simon Bell and Susana Alves and Eva Silveirinha de Oliveira and Affonso Zuin,"Migration and Land Use Change in Europe: A Review", *Living Rev. Landscape Res.* **4**, (2010)

Shaoliang, Y., Ning, W., Peng, L., Qian, W., Fusun, S., Geng, S., Jianzhong, M.2007. Changes in livestock migration patterns in a Tybetan-style agro-pastoral system. Mountain Research and Development, 27(2):138-145.

Stallegger, M. 2008. Management of Natura 2000 habitats. 7150 Depressions on peat substrates of the Rhynchosporion. European Commission, Technical Report 2008 19/24, 23 pp

Vanslembrouck, I., Van Huylenbroeck, G., & Van Meensel, J. (2005). Impact of Agriculture on Rural Tourism: A Hedonic Pricing Approach. *Journal of Agricultural Economics*, *56*(1), 17–30. doi:10.1111/j.1477-9552.2005.tb00120.x

van Oudenhoven , F. J. W., Mijatovic, D., Eyzaguirre, P. B. 2011. Social-ecological indicators of resilience in agrarian and natural landscapes. Management of Environmental Quality: An Internal Journal, 22(2):154-173.

Chapter 3. Differences in the preference of ecosystems and its services: A case study in northern Portugal

Abstract

The importance of determining local resident's preference for ecosystems and their services has become increasingly recognized in conservation and landscape planning. We conducted 120 questionnaires in Portugal, where we explored significant differences in the local's preferences and perception for ecosystems, ecosystem services along an urban-rural gradient. In addition we identify future landscape preference and locals perception of landscape change and possible conservation strategies along the same rurality gradient. Significant differences on the preferences for both ecosystems and ecosystem services were found along the urban-rural gradient. In addition we determined that residents relate conservation value to aesthetic appreciation. Lastly, residents identified the protection of ecosystem services and the local economic development through tourism as primary conservation strategies for the region. These results provide landscape planers with a better understanding of local stakeholders preference for ecosystems and its services, vary spatially and how their perceptions are reflected on the landscape.

Keywords: aesthetic value, ecosystem services, human perception, landscape preference, urban-rural gradient,

3.1. Introduction

Local communities in rural mountain landscapes have lived off the land for centuries, and their perception of value and preference has played a fundamental role in shaping mountain landscapes. According to the European Landscape Convention (ELC) "landscape" means an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors (European Council 2000). This interaction between nature and human activity was essential in the establishment of social-ecological dynamics, provisioning ecosystem services indispensable to human well-being (Calvo-Iglesias et al. 2006; Kaltenborn and Bjerke, 2002). In the last decade, the decline in subsistence agriculture has altered social-ecological dynamics (De Aranzabal et al., 2008), which has been broadly studied at different levels (Deutsch et al. 2003; Martín-López et al. 2011; Michaelidou et al. 2002; Pereira et al. 2005) to fully understand the value and functions provided by these humanized landscapes (Carpenter et al. 2009). Yet, there is still a lack of data on the human preference of ecosystem services at the local scale along a rural gradient.

According to the Millennium Ecosystem Assessment (MA), human well-being is defined by the following dimensions: material minimum for a good life, health, good social relations, security, and freedom of choice (MA 2005). In this context, ecosystem services are related to the benefits that humans obtain from the various ecosystems functions, contributing to human well-being (Costanza et al. 1997; MA 2005). Ecosystems are defined as systems composed of a community of organisms which interact with the environment. Changes in ecosystems impact the provision of ecosystem services affecting human well-being. In Portugal´s Sub-Global Assessment, they revealed that a rural mountain community has undergone positive changes to specific determinants (e.g. security and material) in the last 50 years, while other determinants had decreased (i.e social) impacting human well-being (Pereira et al. 2005).

Both socio-economic and political drivers have contributed to changes in rural inhabitants' perception of value and preference (Calvo-Iglesias et al. 2009; Nikodemus et al. 2005; Roura-Pascual et al. 2005; Westhoek et al. 2006), changing patterns of land use and land management (Calvo-Iglesias et al. 2009; Lasanta-Martínez et al. 2005; Plieninger et al. 2006). One of the main consequences of these social changes is the increasing evidence of abandonment of low intensity farming systems in marginal areas across Europe´s mountain areas (Benayas et al. 2007; Bignal and McCraken 1996; Lasanta-Martínez et al. 2005; MacDonald et al. 2000; Raj Khanal and Watanabe, 2006; Roura-Pascual et al. 2005). These changes have threatened the resilience and sustainability of various ecosystems and their services while contributing to biodiversity change (Busch 2006; Westhoek et al. 2006) and to the erosion of traditional agro-ecological knowledge (Cerqueira et al. 2010; Elbakidze and Angelstam 2007). It has also resulted in more homogeneous landscapes (Bielsa et al. 2005; Höchtl et al. 2005; Roura-Pascual et al. 2005). However, it has opened the opportunity for the restoration of natural processes as well as for conservation projects aimed to promote species, habitats and ecosystem services that are complementary to those provided by cultural landscapes (Navarro and Pereira 2012).

The scale for which humans perceive different landscapes patterns is referred to as the perceptible realm (Goster et al. 2007). It is at this scale that human-environmental interactions take place generating different aesthetic experiences, affecting humans and leading to changes to the landscape and ecosystems. Determining humans' perception and preferences is complex, and humans modify their landscape for ecological, social and economic benefits. However, the task of documenting the human-landscape relationship is complex and multi-level, nonetheless necessary in outlining management policies and conservation projects supporting goals at the local scale (Hunziker et al. 2008; Natori and Chenoweth, 2008; Sayadi et al. 2009; Soliva and Hunziker 2009).

Along the years, a huge body of literature on landscape preference both in terms of conceptual knowledge as well as empirical evidence has been gathered. Several studies have identified landscape preference according to socio-demographic differences (Natori and Chenoweth 2008, Soliva and Hunziker 2009); others have focused on the theory of place attachment (Walker and Ryan 2008). While many, have established a relationship between landscape preference and environmental value orientations (Kaltenborn and Bjerke, 2002; Van den Berg and Koole 2006). Finally, others have studied preference in reference to landscape change (Hunziker et al. 2008) and land uses (Benjamin et al. 2007). Although many studies have addressed issues related to preference and perception of landscape, very few studies have determined preferences for different ecosystems and its services using a non-economic valuation approach (Martín-López et al. 2012).

In this study, we performed a social survey based on a structured questionnaire to assess local inhabitants' preferences of ecosystems, ecosystem services and human well-being in a marginal mountain region of Portugal, undergoing abandonment. Our overarching objective was to determine if there are divergent preferences along a rurality gradient. We also assessed their future landscape preferences and their opinion on which landscape elements should be protected for future conservation strategies.

3.2. Methods

3.2.1. Study area

3.2.1.1. Geography and biophysical conditions

The Aboboreira mountain range is located in northern Portugal shared by the municipalities of Amarante, Baião, and Marco de Canaveses (Fig. 3.1). These three municipalities comprise a total of 91 civil parishes and cover a total area of ca. 680 km^2. This region was chosen for the study as it lies approximately 70 km east of Porto, the second largest city in Portugal, representing a characteristic transitional area from urban to deep rural socio-economy and contrasting land use types. It has undergone some urbanization, primarily those located closest to metropolitan areas, with most of the land use changes commencing in the mid 20th century.

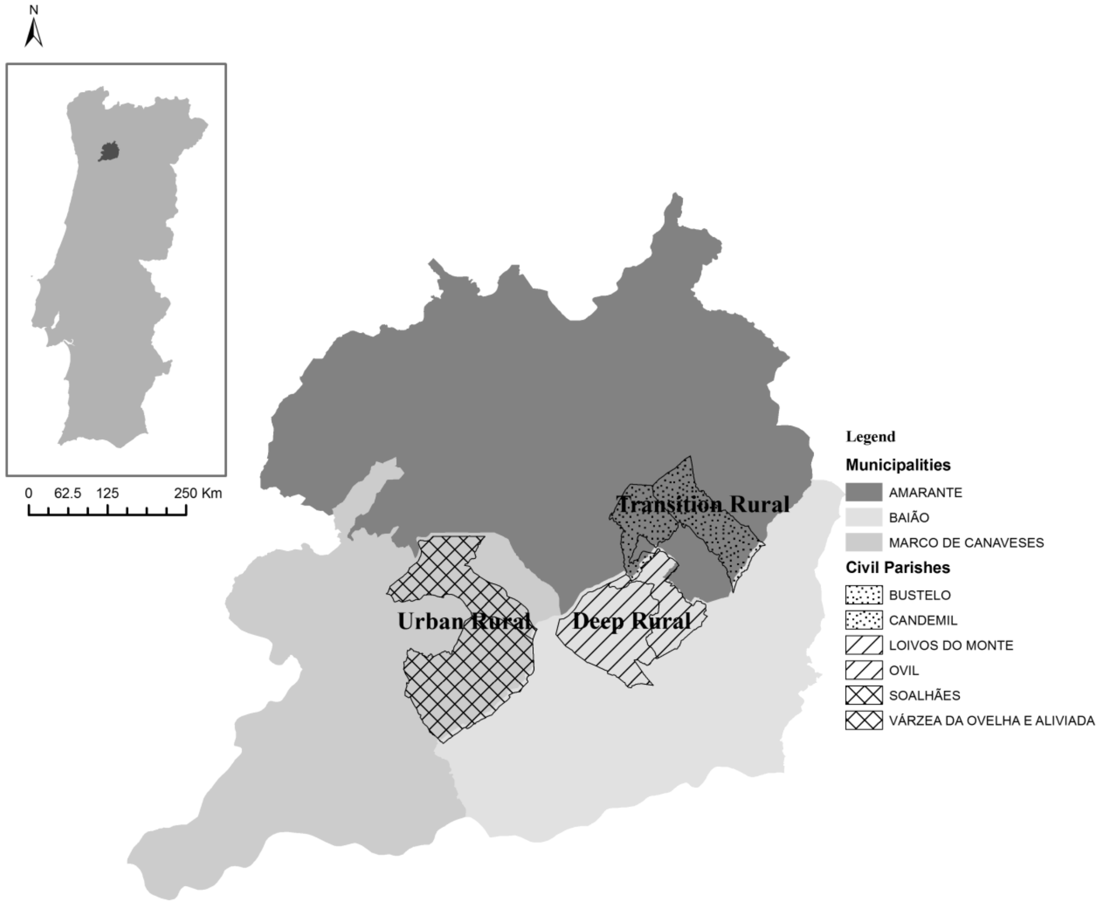

Figure 3.1 Location of the study area (Aboboreira Mountain Range) at the limits of three municipalities in northwest Portugal, and the six surveyed civil parishes organized by socio-economic strata.

The Aboboreira mountain range holds a considerable environmental heterogeneity, with elevation ranging from below 400 m to 1000 m. The region has a temperate climate, with moderate winters and dry cool summers (<40 mm rainfall in the summer period). Annual precipitation ranges from 1600 mm in the valley areas to 2400 mm at higher elevations. Mean annual temperatures range from 16 ºC in lowlands to 12.5 ºC in the higher elevations. Granites are the predominant bedrock types, but soils show marked diversity due to climate, topography and land use.

3.2.1.2. Land use, demography, and recent changes

Typical of many European mountain ranges, the study areas traditional agro-silvo-pastoral systems is characterized by complex mosaics of small patches and many different land uses. Due to high soil fertily, lower valleys where generally dedicated to agriculture. Grazing, on the other hand was mainly along the mountain slopes and in plateaus at higher elevations, where scrubland and grasslands provides more fodder for livestock. Forests are also an integral part of this system, providing various ecosystem services such as timber and fuelwood production. For the most part, the farming system is of a polyculture type (maize, beans and potatoes as dominant crops). In lowlands, vines are found surrounding these cultures, with cabbages and other vegetables growing underneath.

Driven by socio-economic and political drivers, agricultural land abandonment has become evident throughout the region, in particular areas located further away for urbanization. Since the mid 50´s, the lack of job opportunities and of political support led to the emigration of residents to large urban areas and other countries. This trend has continued and recent census in the study area, revealed that between 1991 and 2011, population numbers decreased by 5% and 8% for Amarante and Baião, respectively, and increased 1,9% for Marco de Canaveses (INE 2011). Nonetheless, the percentage of that population represented by an aging population (i.e over the age of 65) has increased by 23%, 15% and 20%, 13,4% above the national percentage (INE 2011). Socio-economic development in Marco de Canaveses has contributed to the increase in population numbers, and a higher representation of age groups between 25-64 (INE 2011). The other municipalities have low representation of a younger generation and mainly host a progressively aging population, making these regions vulnerable to social collapse and land abandonment due to the direct loss of an active population. This demographic depiction is fairly common throughout most rural mountain communities in Portugal and Southern Europe which has resulted in substantial changes to the landscape

According to recent land cover analysis, between 1990 and 2005, the study area suffered an 11% decrease in agricultural area with a subsequent increase in scrubland, particularly in remote and areas located at higher elevations (Honrado 2009). Lower valley regions also endured a decrease in complex small-scale cropland mosaics, which have been replaced by scrubland in some regions and urban expansion. In 2005, agriculture covered, 26.2% of the region, composed of both annual and permanent crops (including meadows and grassland), woodland coverered 20.5% (mainly *Pinus pinaster* stands and deciduous *Quercus robur* and *Q. pyrenaica* forests), 32.8% was occupied by scrubland dominated by gorse, heather or broom (*Erica umbellata*, *Erica cinerea*, *Calluna vulgaris*, *Cytisus striatus*, *Ulex minor* and *Genista triacanthos*), and urban areas accounted for 4.2%. These changes were also followed by a regression in livestock numbers indicating a general trend of abandonment of pastoral activity, further contributing to changes in landscape composition and configuration (Lasanta-Martínez et al. 2005; Rescia et al. 2008).

3.2.2. Sampling design

In this study we intend to determine the preferences of locals along a rurality gradient. According to Selman (2006), landscapes in Europe can be categorized as urban centre, urban fringe, rural of urban and deep rural, each consisting of specific characteristics. The present study area was formerly assumed to be a good representation of the various levels of urbanizations and land cover types (Honrado 2009). This distinction was based on Honrado (2009) preliminary report, delineating various socio-economic degrees and ecological distinctions. For our purposes, the rurality gradient will be defined to include the most profound rural areas to those more urbanized rural regions. For a more indepth distinction of these classes, we identified a number of social and economic indicators to categorize the degree of development of the territory (Fonseca 2000). Data for the socio-economic indicators between 1991 and 2001 were obtained from the Portuguese national statistic office (INE 2011) and are listed in Table.3.1. The stratification allowed the division of complex environmental and socio-economic gradients in operative units, that is, in areas where structures and environmental factors are relatively homogenous. Additionally, in order to evaluate ecological similarities within the municipalities, we used land use map for 2005 (Carta de Ocupação do Solo COS´05) from the Portuguese Geographic Institute (IGEO, 2005). Using the R software (R Development Core Team 2011) we carried out a Partition Around Medoids (PTM) for the various socio-economic indicators and land cover. The algorithm permits a simplification of the various indicators, reducing the heterogeneity of the territory into representative clusters (Kaufman et al. 1990). Based on the various indicators, it was possible to construct a rurality gradient, dividing the territory into 3 distinct area types based on the breaks identified through the partition around medoids algorithm. For our purposes we have designated these areas as deep rural, transition rural and urban-rural.

The designation of deep rural has been widely used in the literature (Antrop 2004; Carvalho Ribeiro and Lovett 2009). Generally consists of an aging population with low residency and little to none commercial or industrial areas. Through the analysis we have designated Baião as largerly characteristic of a deep rural region. Marco de Canaveses consists of mainly urban-rural characteristics, corresponding to the municipality closest to Porto´s outer urban fringe, where secondary (industry) and tertiary (services) sectors are as important or more important that rural activities. In urban-rural areas we have greater residency and commerce with fewer open spaces. Amarante was identified as transition rural, areas which are sparsely populated with greater commercial and industrial development. It is a transitional situation between urban rural and deep rural settings. We then proceeded to randomly select two civil parishes per municipality from the sampling population (Fig. 3.1).

Table 3.1 Demographic and socio-economic variables considered for the stratification of the study area (Source INE 1991 and 2001).

Demographic and socio-economic variables	
Population density (1991-2001) nºkm^2	Unemployment rate (1991 & 2001)
Registered population 1991 and 2001	Active population for the various age groups (2001)
Percentage of population change between 1991-2001	Percentage of economic activity primary sector (2001)
Illiteracy in 2001	Percentage of economic activity secondary sector (2001)
Index of dependent elders (2001)	Percentage of economic activity in the tertiary sector (2001)
Index of aging population (2001)	
Proportion of family accomodation without basic infrastructures (2001)	
Proportion of structures with collection of solid residues (2001)	

3.2.3. Exploratory interviews with local farmers

We started by gathering information from local residents through the use of exploratory interviews. This information was later used to structure the content of the formal survey. A total of six informants were interviewed, one per selected civil parish. The informants included both males and females over 50 years of age, residing in the area for more than 30 years and therefore knowledgeable on environmental and social histories. These semi-structured exploratory interviews consisted of various questions concerning residents' perceptions of value and preferences, the motives for those perceptions and preferences, and future perspectives.

3.2.4. Formal survey

The formal survey was structured into the following sections: (1) assessment of well-being; (2) preference of ecosystems and ranking of most valued ecosystem services; (3) landscape preference and values, as well as preference for future conservation efforts; and (4) perception of landscape change. The survey consisted of both rank order and open questions (Appendix 3.1). In the Appendix (3.1) we provide the full questionnaire, however for the present study we did not consider the full content of the survey.

In the first section of the survey, respondents were asked to rank by order of importance (1-least important to 5-most important) the following dimensions of human well-being (MA 2005): health, social relations, material goods, freedom of choice, and security. For the present study, importance has been defined as something directly or indirectly contributing to their well-being. In addition, determinants were explained to individuals who did not understand the content. The question was asked in the following matter " *By order of importance to you, can you rank the following determinants, health, social relations, material goods, freedom of choice, and security"*.

In the second part of the survey, respondents were asked to rank seven ecosystem types occurring in the region (agricultural field, abandoned farmland, mosaic(agro-forest), scrubland, forest, semi-natural grasslands, and terraces) by order of importance (1-least important to 7-most important). For this part of the survey, importance was defined and explained to the respondents as being something of significant value to them (i.e economic, cultural amongst others). The ecosystem types were selected based on their presence in the study area. Additionally, residents were also asked to rank the different ecosystem types according to their conservation value (representing the ecosystem importance in terms of protection for future conservation) and than for their scenic value. Locals were also asked to rank by order of importance various ecosystem services for both agricultural fields and forests. We highlighted only seven ecosystem services for forests (woodfuel, food, aesthetic appreciation, water supply, leisure, hunting and oxygen (air quality) and three for agricultural fields (food, prevention against soil erosion and aesthetic appreciation). We also left space for other ecosystem services identified by the respondents. The various ecosystem services were selected based on both their priority in terms of conservation (Haines-Young and Potschin 2008) and the information gathered from the exploratory interviews. To facilitate the interpretation of ecosystem services, locals were given a simple definition of ecosystem services and various examples.

In the third section three photographs were provided, each representing a distinct landscape of the study area; traditional cultural landscape, farmland abandonment with scrubland and forest encroachment, and of a modern intensive agricultural landscape (Fig. 3.2). According to various studies, photographs have been proven to be an accurate form of representing real landscapes (Ryan 2005; Walker and Ryan, 2008). Respondents were then asked to rank by order (1-least important to 3-most important) which landscape type they would prefer to see in their civil parish, and why they have selected that particular landscape. The photographs were selected based on information gathered from the exploratory interviews.

Figure 3.2 Representative landscape types in the Aboboreira mountain systems, used in the assessment of landscape preference.

Finally, in the last section, local residents were asked if they were aware of landscape changes in their civil parish in the last two decades, and if they could identify those changes. Participants were then asked their opinion about potential future conservation efforts and asked to identify those elements in the landscape, of conservation value.

3.2.5. Survey administration and participants’ selection

Surveys were conducted by a single surveyor. Each respondent was a willing participant and informed of the content of the survey. All participants were 20 years of age or older and currently residing in the study area. The survey was conducted between June and August of 2009 and obtained a total sample size of 120 respondents, 20 per civil parish (i.e total of 40 surveys per classification; deep rural, transition rural and urban rural). Sampling was conducted using a snowball approach (Pereira et al. 2005), however, there is the risk of bias that should be considered when applying this sampling method, see Lopez et al. (1996).

3.2.6 Data analyses

The analyses focused on determining differences in preference and value along a rurality gradient. Data from the survey were analyzed using SPSS (version 18). We used the non-parametric Friedman Test (significance level $p<0.05$) to determine statistical significant differences (Israel, 2009) in the ranking order of human well-being dimensions, ecosystem types, ecosystem services, and landscape preference along a rurality gradient. We selected the Friedman Test since we are dealing with ordinal data. In order to determine where the significant differences in the rank order occurred along the urban rural gradient, we applied the post hoc Wilcoxon Signed-Rank Test ($p<0.05$) with Bonferroni adjustment ($p<0.017$). We also performed Spearman correlation ($p\leq0.05$) to determine similarities in the ranking order for conservation and aesthetic value attributed for each ecosystem types.

3.3. Results

3.3.1 General characteristics of the sample

More than half of the respondents were males (56.7%) and 31% of the respondents were in the age class of 36-50. A little over one third (36.7%) of the respondents were full- time farmers and only 9.2% had or were near completion of a university degree. Average household income was not taken into consideration as many locals were not receptive to the question, stating that it was a personal matter and refused to answer.

3.3.2. Assessment of human well-being and its determinants

For the first part of the survey, residents were asked to rank by order of importance the determinants of human well-being as defined in the MA (2005). There were no significant differences in the ranking order of human well-being along the urban-rural gradient ($X^2(2)$ =9.367, p=0.08). Respondents generally ranked health (mean= 4.92) as the most important determinant, followed by security (mean=3.26), social relations (mean=3.12), material goods (mean=2.01), and finally freedom of choice and action (mean=1.70).

3.3.3. Preferences for ecosystem types

Agricultural fields, forest, grassland, shrubland and terraces presented similar preference ranking along the urban-rural gradient (Fig. 3.3).

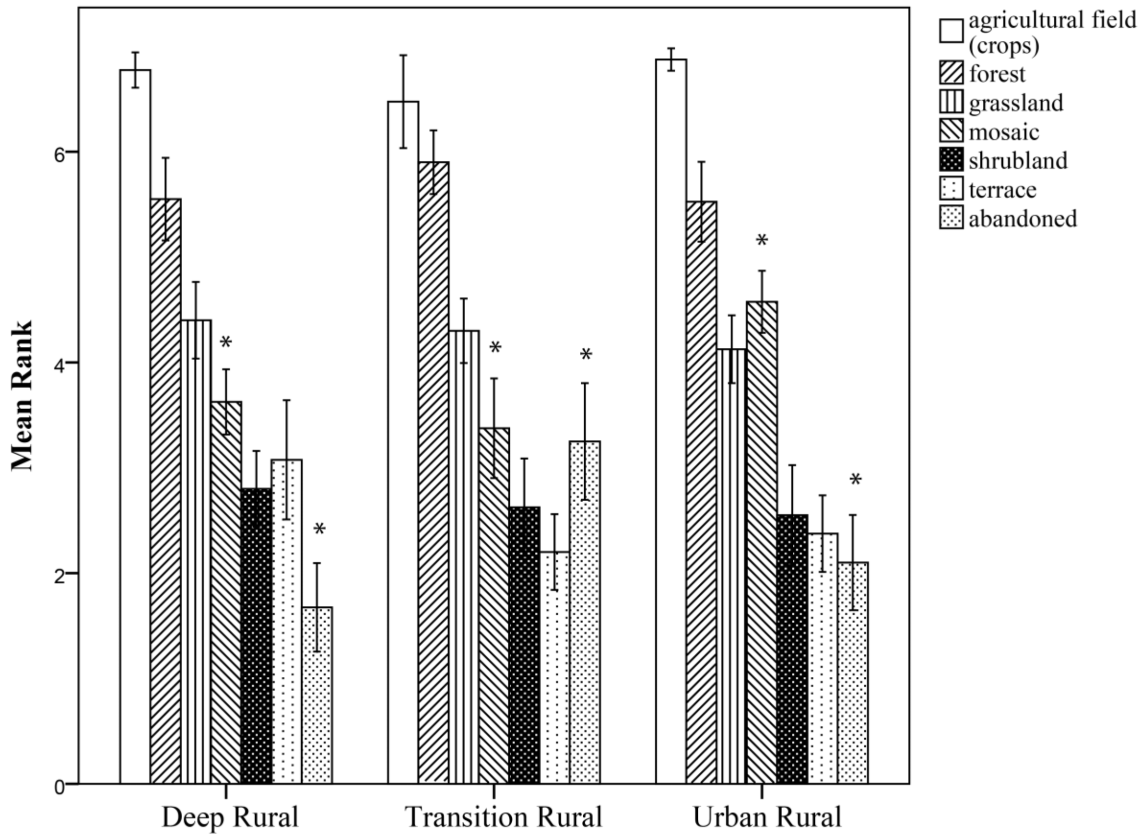

Figure 3.3 Mean rank order of ecosystem types across rurality classes (1-least important to 7-most important). Differences were tested with Friedman Test (p<0.05) followed by post hoc Wilcoxon Signed-Rank Test with Bonferroni adjustment (p<0.017). * shows significant differences along the urban rural gradient for ecosystems. Error bars indicate standard errors.

Generally, agricultural fields (mean=6.71) followed by forest (mean=5.66) were the most preferred ecosystems. On the other hand, abandoned farmland (mean=2.37) and terraces (mean=2.55) were least preferred. Significant differences along the urban-rural gradient for the ranking of abandoned fields ($X^2(2)$ =23.950, p=0.001) and mosaic (agro-forestry) ecosystems ($X^2(2)$ =18.910, p=0.001) (Table.3.2) were found. Transition rural (3.25) parishes ranked abandoned fields higher than both deep rural (1.68) and urban rural (2.10). Also, the ranking of mosaic (agro-forestry) ecosystems was found to be significantly higher in urban rural parishes (4.58) than both deep rural (3.63) and transition rural (3.38) parishes (Fig. 3.3).

Table 3.2 Summary of the statistical analysis for ecosystems, ecosystem services provisioned by forest and landscape preference along an urban-rural gradient.

	Results Friedman Test		Post hoc test					
			d-u		d-t		u-t	
Ecosystems	X^2	p value	z value	p value	z value	p value	z value	p value
Agricultural	1.217	0.544	n.s		n.s		n.s	
Grassland	1.107	0.575	n.s		n.s		n.s	
Mosaic (agro-forestry)	18.919	0.001	-3.587	0.001	n.s		-3.967	0.001
Shrubland	0.944	0.624	n.s		n.s		n.s	
Terrace	2.629	0.269	n.s		n.s		n.s	
Abandoned Farmland	23.95	0.001	n.s		-3.952	0.001	-2.8	0.005
Ecosystem services								
Oxygen	14.656	0.001	n.s		-3.369	0.001	-2.884	0.004
Wood	15.546	0.001	-3.853	0.001	n.s		-3.105	0.002
Food	12.597	0.002	n.s		-3.169	0.002	-2.364	0.016
Aesthetic	5.735	0.057	n.s		n.s		n.s	
water	1.644	0.439	n.s		n.s		n.s	
Leisure	6.457	0.049	-2.393	0.015	n.s		n.s	
Hunting	1.167	0.558	n.s		n.s		n.s	
Landscape Preference								
Traditional cultural	10.658	0.005	-3.503	0.001	n.s		n.s	
Intensive modern	5.313	0.065						
Farmland abandonment	6.271	0.059						

Friedman Test $p<0.05$
Wilcoxon Signed-Rank Test with Bonferonni adjustment $p<0.017$

3.3.4. Preference for ecosystem services

For agricultural fields, respondents recognized only one service, the provision of food, suggesting that respondents are not aware of the multiple ecological and functional role of this ecosystem. In contrast, respondents were able to recognize the various ecosystem services provided by forests. Aesthetic, water and hunting displayed similar rankings along the urban-rural gradient (Fig. 3.4). Largely, inhabitants ranked oxygen (6.33), wood (4.87) and aesthetic (4.38), as the most relevant services provided by forests. Leisure (2.89) and hunting (1.76) received the lowest ranking. In addition, leisure was found to be significantly more important in urban rural parishes compared to deep rural areas (Fig. 3.4).

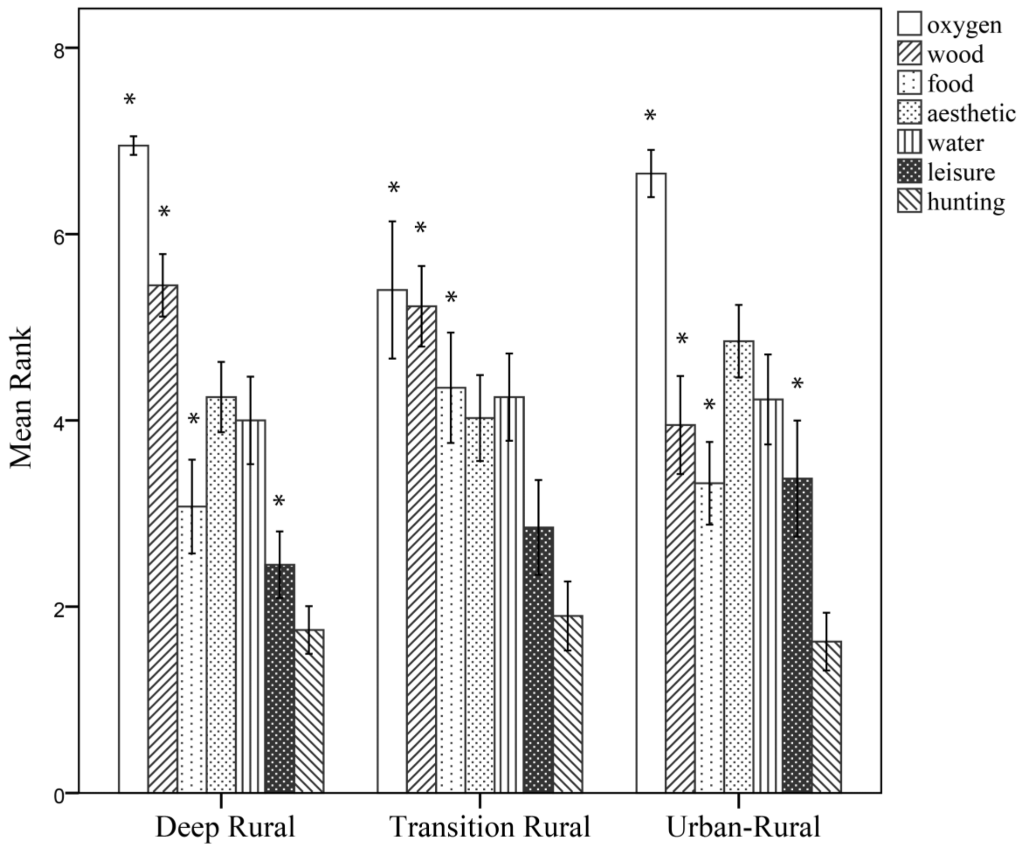

Figure 3.4 Mean rank order for the provision of ecosystem services for forests (1-least important to 7-most important). Differences were tested with Friedman Test ($p<0.05$) followed by post hoc Wilcoxon Signed-Rank Test with Bonferroni adjustment ($p<0.017$). * shows significant differences along the urban rural gradient for ecosystem services. Error bars indicate standard errors.

We found statistically significant differences in the ranking order of leisure, food, air quality and wood along the urban-rural gradient (Table.3.2). The ranking of leisure was significantly higher in urban rural (3.38) than deep rural parishes (2.45) (Fig. 3.4). Food was ranked considerably higher in transition rural (4.35) than in both urban rural (3.33) and deep rural (3.08) parishes, where air quality was ranked most valuable in deep rural parishes (6.95) compared to transition (5.40) and urban rural (6.65) areas. Furthermore, wood was found to be significantly more valued in deep rural and transition rural parishes (5.45, 5.23).

3.3.5. Assessing landscape preferences: contrasting preferences for conservation and scenic value

Forests (c= 6.51; s=6.43) and terraced land (c=6.02; s=5.89) were ranked the most important in terms of both conservation and scenic value (Table 3.3). No significant differences were found along the urban-rural gradient (p=0.435). A strong positive correlation between the rankings for both conservation and aesthetic was found for forest (r=0.946, p<0.001) and terraced land (r=0.890, p<0.001) (Table 3.3).

Table 3.3 Results for Spearman´s rank correlation for ecosystems of conservation and aesthetic value (c-conservation, s-scenic).

Spearman´s rho		agricultural_c	forest_c	terrace_c	abandoned_c	grassland_c	shrubland_c	mosaic_c
agricultural_s	Correlation Coefficient	.285**	-.385**	.302**	-.358**	.331**	-.230*	-.121
	Sig. (2-tailed)	.002	.000	.001	.000	.000	.011	.187
forest_s	Correlation	-.318**	**.946****	-.388**	.388**	-.403**	.170	.003
	Sig. (2-tailed)	.000	.000	.000	.000	.000	.064	.973
terrace_s	Correlation	.349**	-.437**	**.890****	-.212*	.014	-.226*	-.065
	Sig. (2-tailed)	.000	.000	.000	.020	.877	.013	.478
abandoned_s	Correlation	-.253**	.430**	-.319**	.213**	-.386**	-.008	-.135
	Sig. (2-tailed)	.005	.000	.000	.003	.000	.928	.140
grassland_s	Correlation	.065	-.339**	.014	-.287**	.653**	-.017	.013
	Sig. (2-tailed)	.483	.000	.880	.001	.000	.855	.887
shrubland_s	Correlation	-.186*	.221*	-.273**	.141	-.085	.479**	-.276**
	Sig. (2-tailed)	.041	.015	.003	.125	.357	.000	.002
mosaic_s	Correlation	-.060	-.044	-.014	-.173	-.033	-.106	.551**
	Sig. (2-tailed)	.517	.633	.876	.059	.722	.250	.000

**. Correlation is significant at the 0.01 level (2-tailed).

*. Correlation is significant at the 0.05 level (2-tailed).

Overall, 65.8% of the respondents identified cultural landscapes as the preferred landscape. Modern intensive agricultural landscapes and abandoned farmland displayed similar patterns of rank order along the urban-rural gradient (Fig. 3.5). Nonetheless, we found significant differences in the ranking of cultural landscapes ($X^2(2) = 10.658$, *p*=0.005) along the urban-rural gradient (Table. 3.2). Deep rural parishes ranked cultural landscapes higher when compared to urban rural areas (Fig. 3.5).

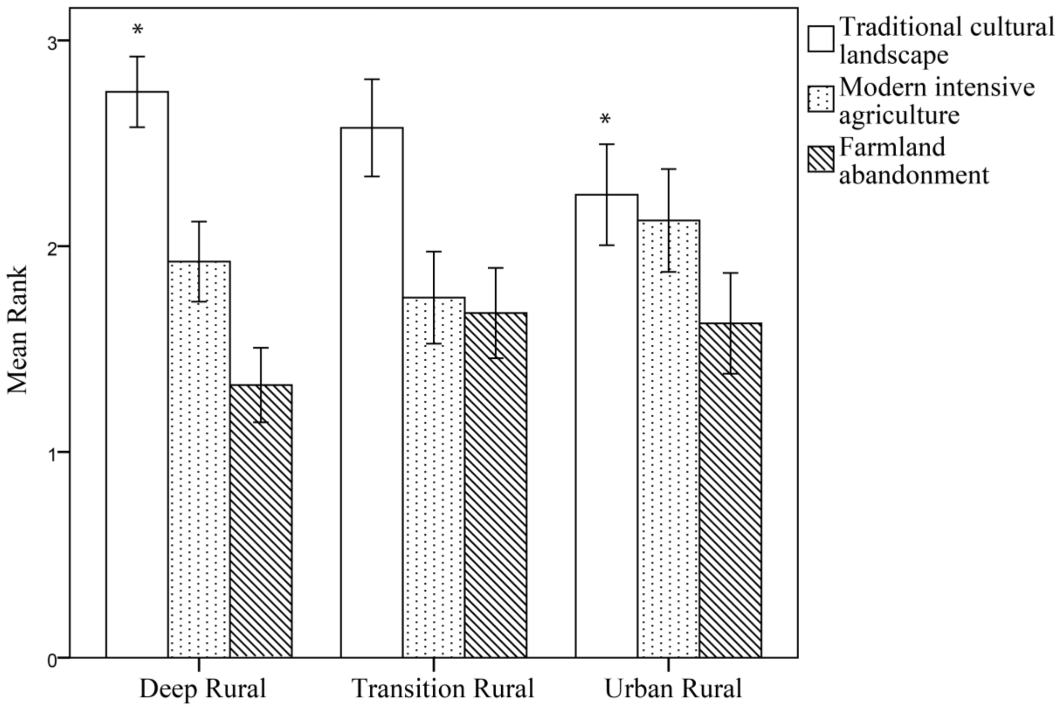

Figure 3.5 Mean rank order of landscape preference (1-least preferred to 3-most preferred). Differences were tested with Friedman Test ($p<0.05$) followed by post hoc Wilcoxon Signed-Rank Test with Bonferroni adjustment ($p<0.017$). * shows significant differences along the urban rural gradient for landscape preference. Error bars indicate standard errors.

The values attributed to the different landscape types (Fig. 3.6) varied. Cultural landscapes were valued for both their scenic (78%) and cultural significance (44%) (Fig. 3.6). In contrast, modern intensive agricultural landscapes were selected for their economic attributes (93%) and abandoned landscapes were recognized for their wilderness (71%) attributes. Wilderness traits were those elements which the respondents referred to as nature, including both flora and fauna. “Everything that is alive is there for a purpose, we see more nature” (interview #24).

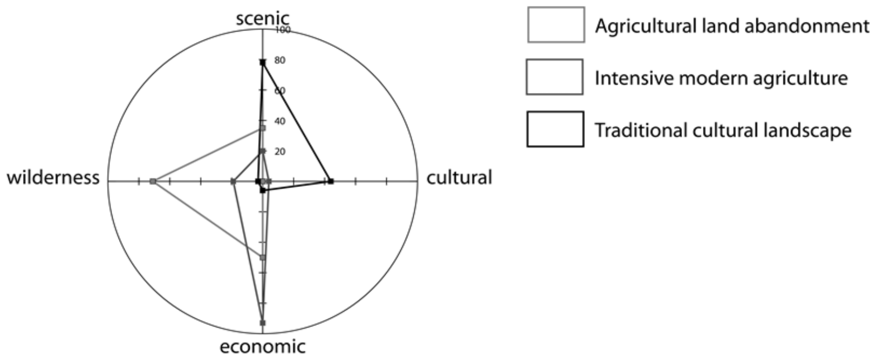

Figure 3.6 Countryside attributes identified by local population for landscape preference.

3.3.6. Perception of landscape change and preference for future conservation efforts

Overall, 75% of the residents inquired perceived landscape changes. However, the perception of landscape change varied along the urban-rural gradient. The three perceptions include an increase in farmland abandonment, an intensification of agriculture and a centralization of cultivation. Residents of deep rural parishes perceived an increase in farmland abandonment (95%) along with an increase in scrubland (97.5%). They also identified that these changes have contributed to an increase in fire occurrence (80%). In contrast, urban-rural parishes state that agriculture has intensified and become more modern (80%) and transition rural parishes have found that cultivation has become more centralized (75%) and forest has steadily increased (83%) along the last twenty years.

The last section of the survey focused on determining what type of strategic conservation planning should be implemented in the study area. The focus of conservation strategies varied along the urban-rural gradient (Fig.3.7). From an ecological viewpoint, deep rural (62.5%) and transition rural (37.5%) residents identified the provision of ecosystem services as the main elements that should be considered for future conservation efforts (Fig. 3.7). Transition rural parishes (37%) also recognized the potential economic benefits to the community through the introduction of tourism. On the other hand, urban rural parishes (45%) identified the regions biodiversity as the potential primary focus of future conservation efforts.

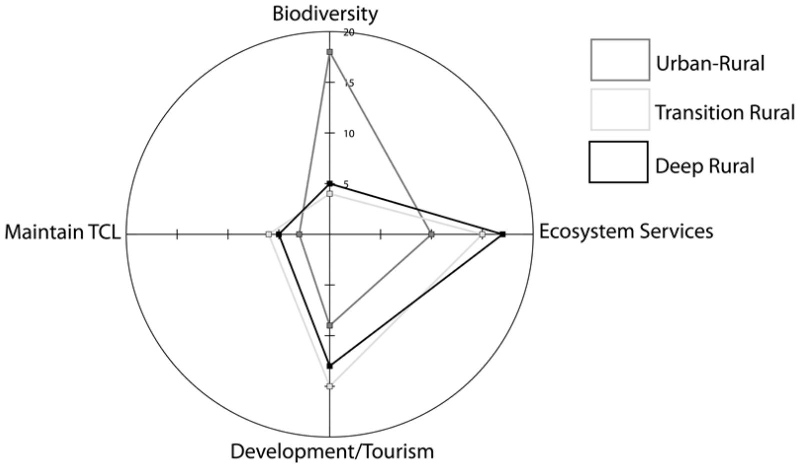

Figure 3.7 Local perception in relation to elements considered in future conservation strategies.

3.4. Discussion

3.4.1. Ecosystems and ecosystem services preference along an urban-rural gradient

Synthesizing different preferences for ecosystems and ecosystem services along an urban rural gradient, provides additional information on those ecosystems and services that are important to humans and how they use and value nature. The connection between nature, human well-being has been widely described. As preference for both ecosystems and ecosystem services change so does the landscape. As a result it has become essential to identify these preferences and how they vary along a rural gradient.

Our study concluded that, indifferent to the urban-rural gradient, residents identified agricultural fields and forests as the most important ecosystems. Significant differences were established for abandoned fields and mosaic ecosystems along the urban-rural gradient, specifically, deep rural parishes ranked abandoned fields lower than transition rural and urban rural areas. Residents of deep rural parishes generally stated that they did not like to see their land abandoned. This perception has also been documented in other mountain communities, such as northern Spain and north-eastern Portugal (Calvo-Iglesias et al. 2009; Pereira et al. 2005). Although residents identified agricultural fields as important, the study areas land cover time series between 1990 and 2005 have showed an 11% decrease in agricultural area, mainly represented at mid to high elevations (i.e deep rural and transitional areas) (Honrado 2009). This decrease can be associated to the out migration of people and the aging population of more rural regions to more urbanized regions, leading to a decreased need of local provisioning services. Generally, terraced land was identified has one of the least important ecosystems, however, was identified as one of the primary ecosystems, for preferred future landscape type. These results are discussed further on.

The preference for agricultural fields and forests is directly and indirectly related to the importance attributed to the various ecosystem services. In the case of agricultural fields, the provision of food was identified as the most important service, indifferent to the rurality gradient. For forests, air quality and wood-fuel were perceived to be the most important services though; significant differences were found along the urban-rural gradient (Fig. 3.4). The identification of these benefits has also been previously identified in past studies (Martín-López et al. 2012). Significant differences were also established in the ranking for leisure and food along the urban-rural gradient, particularly leisure was found of greater value in urban rural parishes compared to deep rural. More urbanized areas sometimes lack leisure spaces, thereby attributing a higher importance to the service. Additionally, leisure (recreation) services are important in contributing to human well-being, mainly health (Radford and James 2013).

Yet, some services supplied by forests, such as hunting and aesthetic services, are ranked similarly along the urban-rural gradient. Overall, hunting was identified as the least important service, suggesting a possible low demand for the service. Contrary to findings in Spain, were hunting was identified as the main service and directly linked to the local's dependence of this particular service, vital in contributing to their well-being (Martín-López et al. 2012). While, we relate the various preferences to the urban-rural gradient, the social differences can be related to various factors (e.g. individuals needs, cultural traditions and accessibility to the service) playing a key role in synthesizing the differences along the urban rural gradient (Carvalho-Ribeiro and Lovett 2011; Martín-López et al. 2012).

3.4.2. Landscape preference and values along an urban-rural gradient

Overall, we determined that generally, locals prefer a more traditional agricultural landscape (Fig. 3.5), yet urban-rural parishes attributed a lower preference in comparison to deep rural areas. Although significant differences were minimal, these differences can be attributed to various aspects such as place attachment and surrounding landscape of upbringing (Kyle et al. 2004; Swanwick 2009; Soliva and Hunziker 2009) as well as human's environmental value orientation (Kaltenborn and Bjerke 2002). According to our results, cultural landscapes were identified as landscapes holding aesthetic and cultural/historical attributes (Fig. 3.6). Modern intensive farming systems were identified as less aesthetically pleasing but on the other hand, economically valued. Whereas, abandoned farmland, were valued for its potential wilderness, through the natural restoration of the vegetation.

Even though, our results identified cultural landscapes as preferred landscape type, they are being abandoned. Determining the perceived importance of terraced landscapes, is a key finding for rural developers, considering that these systems are dependent on human intervention (Henle et al. 2008; Plieninger et al. 2006). If local residents are no longer managing these systems but they remain valuable (e.g. aesthetically) to the community, conservation strategies and planning in the study area need to develop innovative management models to support and protect these systems (Soliva and Hunziker, 2009).

Furthermore, our results highlight that both terraced land and forests hold high conversation and aesthetic value. These findings suggest that locals want to preserve what is aesthetically pleasant to them. In Gobster´s et al. (2007) conceptual model of landscape aesthetics, those landscapes that are perceived to be aesthetically pleasing are stereotypically protected against change. This aesthetic connection ultimately shapes human actions on the landscape. Another concept is that of Hoffenberg (2001) where locals reveal a desire to maintain the landscape connected to historical elements, revealing a degree of attachment to traditional cultural landscapes (Walker and Ryan 2008).

3.4.3. Local identification of landscape changes and conservation values: what it means for future land management plans

Our results also suggest that inhabitants have in the last 20 years, perceived changes to the landscape. Nonetheless this perception of change varied along the urban-rural gradient. Analysis of their responses indicated an increased perception of agricultural abandonment from urban rural to deep rural parishes. Specifically, urban rural populations perceived an intensification of modern agriculture where transition rural parishes acknowledge that agriculture had become centralized and closer to farmers' residence. Comparison of the perception of change with land cover map time series for the period between 1990 and 2005 (Honrado 2009) suggest that residents accurately describe the changes to their surroundings.

Humans have played a fundamental role in shaping our landscapes (Anthrop 2005). The European Landscape Convention has acknowledged that the evolution of the landscape is based on how humans perceive and interact with nature. So, physical changes to the landscape are due to human interaction with ecosystems. Identifying local's perception on conservation planning, specifically, in rural settings has become fundamental in the framework of integrated environmental management. In the present study, participants identified the protection of ecosystem services and the economic development of the region through tourism, targets that should be implemented in future conservation planning. Nonetheless, preferences for different conservation strategies varied along the urban-rural gradient. For example, we found the urban-rural residents were more acquainted with the concept of biodiversity, which can be explained by the combination of both social-economic factors and media communication (Martín-López et al. 2012). Yet, the perception of biodiversity, varied within the urban parishes and other residents. For example, some identified flora and fauna species, where others referred to biodiversity as the whole parish. Another apprehension is the possible connection of human's perception of biodiversity to aesthetic appreciation (Soini and Aakkula 2007).

The development of a conservation strategy that proposes economic development of local communities was referenced, primarily by deep and transition rural parishes (Fig. 3.7). However, this can bring considerable changes to the landscape and we have yet to determine if locals are aware of the potential alterations. For example, in the Vulcano Island in southern Italy, the promotion of tourism led to the replacement of agriculture land for forest (Aretano et al. 2013). Although these land use changes have brought economic relief to the local residents, it resulted in trade-offs between ecosystem services.

Though not mentioned by many, maintaining traditional activities was identified as a positive conservation strategy. Maintaining traditional activities could be a positive ecological, environmental and economic conservation strategy. First, studies show that, agricultural-landscape components are valuable amenities in public preferences and tourists' willingness to pay (Sayadi et al. 2009). Secondly, agricultural landscapes in deep rural areas hold an important component of society´s aesthetic utility and cultural value. Thirdly, maintaining these practices contributes to the resilience of the social-ecological system and contributes to the conservation of biodiversity (Cerqueira et al. 2010; Plieninger et al. 2006).

Although the present study provides a first analysis of preferences along a rural gradient, there were some limitations. Considering that the urban and rural locations were in direct adjacency this could of affected our results by not representing a strong enough variation. The methodology may have introduced a potential bias by pre-defining particular ecosystems and ecosystem services. We must also not exclude the complexity behind the study and the capacity of local's comprehension on the various topics addressed.

3.5. Conclusion

The central aim of this study was to gain greater insights of diverging preferences towards ecosystems and ecosystem services and perception of landscape change along an urban-rural gradient. Overall, we determined that the perception and valuation of certain ecosystems varied along an urban-rural gradient as well as perception of landscape change. Additionally, aesthetic value influenced conservation preference. By identifying aesthetic and conservation preference for ecosystems and the demand for ecosystem services we can attempt to reconcile both within a management plan (Gobster et al. 2007). However, it might be argued that due to scale mismatches between the perceptible realm and the scales at which ecosystem services are provided and delivered and the ways in which an area of land is perceived by people many not run parallel to the regions full ecological importance. This raises enormous challenges for management and planning. Our findings highlight the presence of divergent preferences along a rural gradient, contributing to the on-going need to recognize the spatial demand of particular services and plan for multifunctional landscapes (Soliva and Hunziker 2009; Surová and Pinto-Correia 2008).

Although the present study provides some insights on local preferences, we must not discard its limitations, such as the small sample size. A deeper engagement with local populations through workshops would contribute to a greater understanding regarding preference and perception, supporting the data obtained from the surveys.

The on-going land abandonment throughout much of Europe´s rural mountain areas has led to the natural restoration of ecosystems or rewilding. Although, much scepticism can be found around the topic of rewilding, it is thought to bring environmental, economic and social benefits, specifically to rural mountain communities (Navarro and Pereira 2012). While, determining locals perception on this topic was not an objective of the present study, our results show that forests are valuable to locals we therefore suggest future research to determine locals view on the topic of rewilding.

3.6. References

Aguiar, C., Rodrigues, O., Azevedo, J., Domingos, T. (2009). Montanha. In H. M. Pereira, T. Domingos, L. Vicente, V. Proença (Eds), Ecossistemas e Bem-Estar Humano: *Avaliação para Portugal do Millenium Ecosystem Assessment* (pp. 295-339). Fundação da Faculdade de Ciências da U.L e Escolar Editora.

Antrop, M.,2005. Why landscapes of the past are important for the future. Rural Landscapes: past processes and future strategies 70, pp. 21–34.

Aretano, R., Petrosillo, I., Zaccarelli, N., Semeraro, T. and Zurlini, G., 2013. People perception of landscape change effects on ecosystem services in small Mediterranean islands: A combination of subjective and objective assessments. Landscape and Urban Planning 112, pp. 63–73. doi:10.1016/j.landurbplan.2012.12.010

Bartlett, J. E., Kotrlik, J. W. and Higgins, C. C., 2001. Organizational Research: Determining Aprropriate Sample Size in Survey Research. Information Technology, Learning, and Performance Journal 19, pp.43-50.

Bassi, M. and Tache, B., 2011. The community conserved landscaped of the Borana Oromo, Ethiopia. Management of Environmental Quality 22, pp. 174-186.

Benayas, J. M., Martins, A., Nicolau, J. M. and Schulz, J. J., 2007. Abandonment of agricultural land: an overview of drivers and consequences. CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources 2, pp. 1-14.

Benjamin, K., Bouchard, A. and Domon, G., 2007. Abandoned farmlands as components of rural landscapes: An analysis of perceptions and representations. Landscape and Urban Planning 83(4), 228–244. doi:10.1016/j.landurbplan.2007.04.009

Berkes, F., 2004. Repensando la Conservación Basada en Comunidades. Conservation Biology 18(3), pp. 621–630. doi:10.1111/j.1523-1739.2004.00077.x

Bielsa, I., Pons, X. and Bunce, B., 2005. Agricultural Abandonment in the North Eastern Iberian Peninsula: The Use of Basic Landscape Metrics to Support Planning. Journal of Environmental Planning and Management 48(1), pp. 85–102. doi:10.1080/0964056042000308166

Bignal, E. M. and McCracken, D. J., 1996. Low-intensity farming systems in the conservation of the countryside. Journal of Applied Ecology 33(1), pp. 413-424.

Busch, G., 2006. Future European agricultural landscapes—What can we learn from existing quantitative land use scenario studies? Scenario-Based Studies of Future Land Use in Europe 114(1), pp. 121–140. doi:10.1016/j.agee.2005.11.007

Calvo-Iglesias, M. Silvia, Crecente-Maseda, R. and Fra-Paleo, U., 2006. Exploring farmer's knowledge as a source of information on past and present cultural landscapes: A case study from NW Spain. Landscape and Urban Planning 78(4), pp. 334–343. doi:10.1016/j.landurbplan.2005.11.003

Calvo-Iglesias, María Silvia, Fra-Paleo, U. and Diaz-Varela, R. A., 2009. Changes in farming system and population as drivers of land cover and landscape dynamics: The case of enclosed and semi-openfield systems in Northern Galicia (Spain). Landscape and Urban Planning 90(3–4), pp. 168–177. doi:10.1016/j.landurbplan.2008.10.025

Carpenter, S. R., Mooney, H. A., Agard, J., Capistrano, D., DeFries, R. S., Díaz, S., … Whyte, A., 2009. Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. Proceedings of the National Academy of Sciences 106(5), pp. 1305–1312.

Carta de Ocupação do Solo. 2005. COS´2005

Carvalho-Ribeiro, S. M. and Lovett, A., 2011. Is an attractive forest also considered well managed? Public preferences for forest cover and stand structure across a rural/urban gradient in northern Portugal. Forest Policy and Economics 13(1), pp. 46–54. doi:10.1016/j.forpol.2010.09.003

Cerqueira, Y. C., Araújo, C., Vincente, J., Pereira, H. M. and Honrado, J., 2010. Ecological and cultural consequences of agricultural abandonment in the Peneda-Gerês National Park (Portugal). In: T. Evelpidou, T. Figueiredo, F. Mauro, V. Tecim, A. Vassilopoulos. (Eds), *Natural Heritage from East to West: Case studies from 6 EU countries* (pp. 175-183). London, New York.

Costanza, R., d´Arge, De Groot, R., Farber, S., Grasso, M., Hannon, B. et al., 1997. The value of the world´s ecosystem services and natural capital. Nature 387(1), pp. 253-260.

De Aranzabal, I., Schmitz, M. F., Aguilera, P. and Pineda, F. D., 2008. Modelling of landscape changes derived from the dynamics of socio-ecological systems: A case of study in a semiarid Mediterranean landscape. Ecological Indicators 8(5), pp. 672–685. doi:10.1016/j.ecolind.2007.11.003

Deutsch, L., Folke, C. and Skånberg, K., 2003. The critical natural capital of ecosystem performance as insurance for human well-being. Identifying Critical Natural Capital 44(2–3), pp. 205–217.

EEA, European Environmental Agency. (2004). High nature value farmland: characteristics, trends and policy challenges. Report N 1/2004, Copenhagen.

Elbakidze, M. and Angelstam, P., 2007. Implementing sustainable forest management in Ukraine's Carpathian Mountains: The role of traditional village systems. Traditional knowledge, cultural heritage and sustainable forest management 249(1–2), pp. 28–38. doi:10.1016/j.foreco.2007.04.003

European Council 2000, European Landscape Convention, Council of Europe, Strasbourg,France, viewed 13 June 2006, <http://www.coe.int/t/e/Cultural_Co-operation/Environment/Landscape/>

Fonseca, P.A.L., 2000. Indices de desenvolvimento concelho. Volume II. 2. Quadrimestre de 2002. Instituto Nacional de Estatistica, Lisbon, pp. 1–34.

Gobster, P. H., Nassauer, J. I., Daniel., T. C., Fry, G. 2007. The shared landscape: what does aesthetics have to do with ecology? Landscape Ecology, 22:959-72.

Haines-Young, R., & Potschin, M. (2008). England's terrestrial ecosystem services and the rationale for an ecosystem approach – Overview report to Defra [Project Code NR0107]. Available from http://www.ecosystemservices.org.uk/docs/NR0107 Overview%20report 080108.pdf Accessed 02.12.08.

Henle, K., Alard, D., Clitherow, J., Cobb, P., Firbank, L., Kull, T., … Young, J., 2008. Identifying and managing the conflicts between agriculture and biodiversity conservation in Europe–A review. Special Section: Problems and Prospects of Grassland Agroecosystems in Western China 124(1–2), pp. 60–71. doi:10.1016/j.agee.2007.09.005

Höchtl, F., Lehringer, S. and Konold, W., 2005. "Wilderness": what it means when it becomes a reality—a case study from the southwestern Alps. Rural Landscapes: past processes and future strategies 70(1–2), pp. 85–95. doi:10.1016/j.landurbplan.2003.10.006

Hoffenberg, P. H. , 2001. Landscape, Memory and the Australian War Experience, 1915–18. Journal of Contemporary History 36(1), pp. 111–131.

Honrado, J. (coord.), 2009. *O património natural e cultural como factor de desenvolvimento e competitividade territoriais no Baixo Tâmega.* Final report, Faculdade de Ciências da Universidade do Porto.

Hunziker, M., Felber, P., Gehring, K., Buchecker, M., Bauer, N. and Kienast, F., 2008. Evaluation of Landscape Change by Different Social Groups. Mountain Research and Development 28(2), pp. 140–147. doi:10.1659/mrd.0952

IGEO, 2005. Carta de ocupaça˜o do solo (COS' 05). Instituto Geografico de Portugal

Instituto Nacional de Estatística (INE). 1990. Recenseamento geral da agricultural. INE, Lisbon Portugal.

Instituto Nacional de Estatística (INE). 2009. Recenseamento geral da agricultural. INE, Lisbon Portugal.

Instituto Nacional de Estatistíca (INE). 2001. Recenseamento geral da população. INE, Lisbon Portugal.

Israel, D., 2009. Data Analysis in Business Research: A Step-By-Step Nonparametric Approach. Sage Publications Pvt. Ltd.

Kaltenborn, B. P. and Bjerke, T., 2002. Associations between environmental value orientations and landscape preferences. Landscape and Urban Planning 59(1), pp. 1–11. doi:10.1016/S0169-2046(01)00243-2

Kaufman, L. and Rousseeuw, P.J., 1990. Finding Groups in Data: An Introduction to Cluster Analysis. John Wiley & Sons, New York.

Khanal, N. R. and Watanabe, T., 2006. Abandonment of agricultural land and its consequences. Mountain Research and Development 26(1), pp. 32-40.

Kyle, G. T., Mowen, A. J. and Tarrant, M., 2004. Linking place preferences with place meaning: An examination of the relationship between place motivation and place attachment. Journal of Environmental Psychology 24(4), pp. 439–454. doi:10.1016/j.jenvp.2004.11.001

Lasanta, Teodoro, Arnáez, J., Oserín, M. and Ortigosa, L. M., 2001. Marginal Lands and Erosion in Terraced Fields in the Mediterranean Mountains. Mountain Research and Development 21(1), pp. 69–76. doi:10.1659/0276-4741(2001)021[0069:MLAEIT]2.0.CO;2

Lasanta-Martínez, T., Vicente-Serrano, S. M. and Cuadrat-Prats, J. M., 2005. Mountain Mediterranean landscape evolution caused by the abandonment of traditional primary activities: a study of the Spanish Central Pyrenees. Applied Geography 25(1), pp. 47–65. doi:10.1016/j.apgeog.2004.11.001

MacDonald, D., Crabtree, J. ., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., … Gibon, A., 2000. Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. Journal of Environmental Management 59(1), pp. 47–69. doi:10.1006/jema.1999.0335

Martín-López, B., García-Llorente, M., Palomo, I. and Montes, C., 2011. The conservation against development paradigm in protected areas: Valuation of ecosystem services in the Doñana social–ecological system (southwestern Spain). Ecological Economics 70(8), pp. 1481–1491. doi:10.1016/j.ecolecon.2011.03.009

Martín-López, B., Iniesta-Arandia, I., García-Llorente, M., Palomo, I., Casado-Arzuaga, I., Amo, D. G. D., … Montes, C., 2012. Uncovering Ecosystem Service Bundles through Social Preferences. PLoS ONE 7(6), e38970. doi:10.1371/journal.pone.0038970

Michaelidou, M., Decker, D. J. and Lassoie, J. P., 2002. The Interdependence of Ecosystem and Community Viability: A Theoretical Framework to Guide Research and Application. Society & Natural Resources 15(7), pp. 599–616. doi:10.1080/08941920290069218

Millennium Ecosystem Assessment (2005). Ecosystems and Human well-being: Biodiversity Synthesis. World Resources Institute, Washington, DC.

Natori, Y. and Chenoweth, R., 2008. Differences in rural landscape perceptions and preferences between farmers and naturalists. Journal of Environmental Psychology 28(3), pp. 250–267. doi:10.1016/j.jenvp.2008.02.002

Navarro, L. and Pereira, H., 2012. Rewilding Abandoned Landscapes in Europe. Ecosystems, 15(6), pp. 900–912. doi:10.1007/s10021-012-9558-7

Nikodemus, O., Bell, S., Grine, I. and Liepins, I., 2005. The impact of economic, social and political factors on the landscape structure of the Vidzeme Uplands in Latvia. Rural Landscapes: past processes and future strategies 70, pp. 57–67. doi:10.1016/j.landurbplan.2003.10.005

Pereira, E., Queiroz, C., Pereira, H. M. and Vicente, L., 2005. Ecosystem Services and Human Well-Being: a Participatory Study in a Mountain Community in Portugal. Ecology and Society, 10(2).

Perrings, C., Duraiappah, A., Larigauderie, A. and Mooney, H., 2011. The Biodiversity and Ecosystem Services Science-Policy Interface. Science 331(6021), pp. 1139–1140.

Plieninger, T., Höchtl, F. and Spek, T., 2006. Traditional land-use and nature conservation in European rural landscapes. European Rural Landscapes: Land-use, biodiversity, and conservation 9(4), pp. 317–321.

R Development Core Team., 2011. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-

900051-07-0, [online] URL: http://www.R-project.org/.

Radford, K. G. and James, P., 2013. Changes in the value of ecosystem services along a rural–urban gradient: A case study of Greater Manchester, UK. Special Issue: Urban Ecosystem Services 109(1), pp. 117–127. doi:10.1016/j.landurbplan.2012.10.007

Rescia, A. J., Pons, A., Lomba, I., Esteban, C. and Dover, J. W., 2008. Reformulating the social–ecological system in a cultural rural mountain landscape in the Picos de Europa region (northern Spain). Landscape and Urban Planning 88(1), pp. 23–33.

Roura-Pascual, N., Pons, P., Etienne, M. and Lambert, B., 2005. Transformation of a Rural Landscape in the Eastern Pyrenees Between 1953 and 2000. Mountain Research and Development 25(3), pp. 252–261.

Ryan, R. L., 2005. Exploring the Effects of Environmental Experience on Attachment to Urban Natural Areas. Environment and Behavior 37(1), pp. 3–42.

Sayadi, S., González-Roa, M. C. and Calatrava-Requena, J., 2009. Public preferences for landscape features: The case of agricultural landscape in mountainous Mediterranean areas. Land Use Policy 26(2), pp. 334–344.

Soini, K. and Aakkula, J., 2007. Framing the biodiversity of agricultural landscape: The essence of local conceptions and constructions. Land Use Policy 24(2), pp. 311–321.

Soliva, R. and Hunziker, M., 2009. How do biodiversity and conservation values relate to landscape preferences? A case study from the Swiss Alps. Biodiversity and Conservation 18(9), pp. 2483–2507.

Surová, D. and Pinto-correia, T., 2008. Landscape preferences in the cork oak Montado region of Alentejo, southern Portugal: Searching for valuable landscape characteristics for different user groups. Landscape Research 33(3), pp. 311–330.

Swanwick, C., 2009. Society’s attitudes to and preferences for land and landscape. Land Use Futures, 26, Supplement 1(0), S62–S75. doi:10.1016/j.landusepol.2009.08.025

Swiderska, K., 2003. Integrating local values in natural resource assessment: a review of assessment tools. IEED Natural Resources Group.

TEEB. 2010. A Quick Guide to the Economics of Ecosystems and Biodiversity for Local and Regional Policy Makers. [online] URL: http://www.teebweb.org.

Van den Berg, A. E. and Koole, S. L., 2006. New wilderness in the Netherlands: An investigation of visual preferences for nature development landscapes. Landscape and Urban Planning 78(4), pp. 362–372.

Walker, A. J. and Ryan, R. L., 2008. Place attachment and landscape preservation in rural New England: A Maine case study. Landscape and Urban Planning 86(2), pp. 141–152.

Westhoek, H. J., Van den Berg, M. and Bakkes, J. A., 2006. Scenario development to explore the future of Europe's rural areas. Scenario-Based Studies of Future Land Use in Europe 114(1), pp. 7–20.

Appendix

Questionnaire applied in the Aboboreira Mountain Range.

Percepção de Valor nas Áreas Serranas do Baixo Tâmega

Freguesia/Aldeia/Lugar:__________________________ Data:_______________

Entrevistador:____

Horário de Inicio:_______Horário Terminado:________Tempo Total:______

Nome Completo:____

Residente há quanto tempo:____

Profissão:____________________________________Idade:_________________

Questionário

Secção 1. Percepção de Valor e de Critérios de Valor

1.1. Conceito de "bem-estar"

1. O bem-estar humano para si é definido como (Indique por ordem qual é o mais importante para si 1 sendo o mais importante e 5 o menos importante).

a) Saúde ____
b) Segurança ____
c) Social- Convívio ____
d) Material- Bens propriedades (para ter uma boa vida ____
e) Liberdade de escolha ____

1.2. Apreço pelo lugar de residência

2. Gosta de viver na sua terra? **SIM** □ **NÃO**□

Porquê? ____

1.3. Valoração de ecossistemas (florestas)

3. Na sua opinião, as florestas são importantes porquê? Indique por ordem qual é o mais importante para si.

a) Fornecem lenha ___
b) Fornecem alimentos __
c) Fornecem ou contribuem para uma paisagem bonita ___
d) Fornecem água___
e) Laze ___
f) Caça ___
g) Oxigénio ___
h) Outro (________________________)___
i) Não são importantes __

1.4. Valoração de ecossistemas (campos)

4. Na sua opinião, os campos são importantes porquê? Indique por ordem qual é o mais importante para si?

a) Fornecem alimentos. ___
b) Embelezem a paisagem. ___
c) Contribuem para evitar erosão do solo.___
d) Outro (_________________________)__
e) Não são importantes. ___

1.5. Valoração de ecossistemas (importância global)

5. Quais são mais importantes para si? (1 sendo o mais importante e 7 menos importante)

a) Campos agrícolas ____
b) Florestas ___
c) Socalcos □____

d) Baldios ____
e) Lameiros ___
f) Matos____
g) Prados
h) Outro
i) Nenhum

1.6. Valoração de ecossistemas (valor cénico da paisagem)

6. Na sua opinião, o que é que contribui mais para a beleza da paisagem? (1 sendo o mais importante e 7 menos importante).

a) Campos agrícolas ____
b) Florestas ____
c) Socalcos ____
d) Baldios ____
e) Lameiros ____
f) Matos ____
g) Prados ____
h) Outro (__________________________) ____
i) Nenhum ____

1.7. Valoração de ecossistemas (necessidade de protecção)

7. Por ordem de importância (1 sendo o mais importante e 7 menos importante), **quais destas paisagens devem ser protegidas**:

a) Campos agrícolas ____
b) Florestas ____
c) Socalcos ___
d) Baldios ____
e) Lameiros ____
f) Matos ____
g) Prados ___
h) Outro (__________________________)____
i) ____

1.8. Valoração de espécies (valor global)

8. Qual destas árvores é mais importante para si? (1 sendo o mais importante e 6 menos importante)

a) Freixo____
b) Carvalho ____
c) Sobreiro ____
d) Castanheiro ____
e) Pinheiro ____
f) Eucalipto ____

1.9. Valoração de espécies (valor cénico da paisagem)

9. Quais espécies mais contribuem para a beleza da paisagem? (1 sendo o mais bonito e 6 menos bonito)

a) Freixo ____
b) Carvalho ____
c) Sobreiro ____
d) Castanheiro ____
e) Pinheiro ____
f) Eucalipto ____

1.10. Valoração de paisagens (valor global)

10. Qual destas paisagens prefere? (ordene as fotos)

Foto____
Indique os motivos:

a.) Valor económico potencial

b.) Beleza (qualidade estética)

c.) Sensação de conforto e segurança

d.) Familiaridade (reconhecimento)

e.) Valor cultural e histórico

f.) Outro (_________________________________)

Foto____
Indique os motivos:

a.) Valor económico potencial

b.) Beleza (qualidade estética)

c.) Sensação de conforto e segurança

d.) Familiaridade (reconhecimento)

e.) Valor cultural e histórico

f.) Outro (_________________________________)

Foto____
Indique os motivos:

a.) Valor económico potencial

b.) Beleza (qualidade estética)

c.) Sensação de conforto e segurança

d.) Familiaridade (reconhecimento)

e.) Valor cultural e histórico

f.) Outro (_________________________________)

Secção 2. Percepção de Alterações do Uso do Solo e Seus Promotores

2.1. Percepção de alterações na paisagem (global)

11. Nos últimos 20 anos a paisagem (terra) mudou? **SIM** □ **NÃO** □ (Se sim identifique).

a.) Aumento em campos abandonados.

b.) Diminuição de campos abandonados.

c.) Aumento de florestas.

d.) Diminuição de florestas.

e.) Aumento de ocorrência de fogos.

f.) Diminuição de ocorrência de fogos.

g.) Intensificação da agricultura.

h.) Abandono de praticas agrícolas.

i.) Outro. Indique qual: ___________________

<u>2. Percepção de alterações na floresta e os seus promotores</u>

12. Nos últimos 20 anos a floresta aumentou? **SIM** □ **NÃO** □ (Se sim identifique as razões).

a.) Aumento das espécies exóticas (pinheiro e eucalipto) que não são cortadas.

b.) As pessoas emigraram.

c.) A ocorrência de fogos diminuiu.

d.) Diminuição no cultivo e aumento no abandono de campos agrícolas.

e.) O cultivo agora e centralizado e mais pequeno.

f.) Outro: Identifique____

<u>2.2. Percepção de alterações das espécies</u>

13. Para cada espécie, **indique** se lhe parece que está a aumentar, diminuir ou se mantém igual.

a.) Freixo **Aumentar** **Diminuir** **Igual**

b.) Carvalho **Aumentar** **Diminuir** **Igual**

c.) Sobreiro **Aumentar** **Diminuir** **Igual**

d.) Castanheiro **Aumentar** **Diminuir** **Igual**

e.) Pinheiro **Aumentar** **Diminuir** **Igual**

f.) Eucalipto **Aumentar** **Diminuir** **Igual**

2.3. Fogo: consequência de abandono de campos agrícolas

14. Nestes últimos 20 anos acha que os fogos são

a.) mais frequentes **SIM** **NÃO**

b.) intensos e de maior extensão **SIM** **NÃO**

c.) igual **SIM** **NÃO**

d.) outro:_________ **SIM** **NÃO**

E **porquê**?

____

____

2.4. Percepção de alterações futuras a terra cultivada e os seus promotores

15. A terra cultivada vai diminuir no futuro? SIM □ NÃO □ (Se sim identifique as razões).

a.) a floresta esta a aumentar

b.) subsídios são baixos (CAP)

c.) intensificação do uso do solo

d.) não a quem cultive a terra

e.) Outro: Identifique_______________

16. Tem terra para cultivar (passado e presente)? **SIM □ NÃO □ (se respondeu não passa para pergunta 19)**

17. Nos últimos **20 anos** houve alterações no uso do solo nos seus campos?

SIM □ NÃO □

18. Se sim identifique as alterações:

a.) abandono □

b.) modificações no que é cultivado (indique o que era cultivado e o que cultivado agora)____

c.) clareiras **aumento □ diminuição □**

d.) outro:____

19. As terras que já não são cultivadas têm alguma função? SIM □ NÃO □

a) pastorícia

b) recolha de matos/lenha

c) outro:_____________________

Secção 3. Percepção de Impactos sobre o Valor da Paisagem e o Conhecimento Ecológico Tradicional (TEK)

3.1. O passado e futuro das terras cultivadas

20. Neste momento cultiva a terra? SIM □ NÃO □ (Se cultiva por favor indique o que é cultivado): **(se respondeu NÃO passe para pergunta 25)**

a.) cereais

b.) vinha

c.) horta (feijão, cenoura, couve etc..)

d.) pastorícia

e.) outro:_________________________

21. Se **SIM** identifique se a forma de cultivar a terra é: **Moderna □Tradicional □**

22. Acha que **no futuro** a sua terra vai continuar a ser cultivada? **SIM □ NÃO □**

23. A terra continuara a ser cultivada por quem?

a.) Próprio (proprietário) □

b.) Familiar □ Identifique_______

c.) Outro:__________________

24. Se NÃO, porquê?

a.) Não há ninguém para cultivar a terra.

b.) Já não compensa cultivar a terra.

c.) Já não é preciso cultivar a terra.

d.) Outro:________________________

25. Quais são as **alterações** que consegue identificar na paisagem com a utilização de praticas de cultivo moderno? (**pode escolher mais que uma**)

a.) campos mais largos

b.) agricultura mais intensa (exemplo: monocultura)

c.) modificação de sistema de rega

d.) infrastructuras (muros e sebes)

e.) aumento em acessos

f.) diminuição de vegetação (matos, giestal e tojo)

g.) outros:____

3. Promotores de Conhecimento Ecológico Tradicional (TEK)

26. Recebe subsídios? **SIM** □ **NÃO** □ Identifique o tipo de subsídio:
____

27. Acha que subsídios são a solução para preservar as tradições agrícolas?

SIM□ NÃO □ Se não identifique a/as razões?

____

____

3. Valorização de praticas agrícolas tradicionais e percepção de valor cultural

28. Trabalhar no campo ainda compensa? **SIM □ NÃO □**

E porquê? ____

29. Preferia (valoriza) cultivar a terra de forma **moderna** ou **tradicional**?

Moderna □

Tradicional □

30. Hoje, quais são as principais diferencias nas **praticas agrícolas tradicionais**?

a.) introdução de maquinas (tractores etc…)

b.) agricultura mais intensiva (maior produção) e centralizada c.) químicos

d.) alterações nas culturas

e.) custos (rações e outros)

f.) outros:____

31. As infra-estraturas (muro de pedra) presentes nos campos agrícolas têm algum valor para si?
SIM □ **NÃO**□ E porquê?

a.) tem valor cénico

b.) tem uma função (segurar o gado)

c.) outro:___________________________

3. O futuro do Conhecimento Ecológico Tradicional

32. O que acha que vai acontecer as tradições/práticas agrícolas?

a.) vão ser descontinuadas □

b.) vão continuar a ser praticadas pelo

-os jovens □ pelos os antigos □ pelos os dois □

c.) vão ser substituídas por tecnologia moderna □

d.) não sei □

Secção 4. Expectativas para o Futuro

4.1. Conhecimento e a importância da conservação da paisagem

33. A conservação das terras é **importante** porque protege (pode responder mais que uma):

a.) florestas

b.) campos (agricultura)

c.) animais/plantas

d.) o solo/a terra

e.) outro:____________

f.) a conservação das terras não é importante

34. Reconhece a importância da **Serra da Aboboreira**? **SIM** □ **NÃO** □ (Se sim identifique as razões).

a.) Preservar a biodiversidade

b.) Bens fornecidos pelo o ecossistema

c.) Turismo

d.) Ciência

e.) Direitos de propriedade

f.) Desenvolvimento da área

g.) Aumento de emprego

h.) Outro. Indique qual:_______________

35. Na sua opinião a **Serra da Abobereira** deveria ser classificada como área protegida? **SIM** □ **NÃO** □

Porquê?____
____

36. Acha que a sua freguesia devera ser incluída na area protegida? **SIM** □ **NÃO**□

Porquê?____

Chapter 4. Assessing ecosystem services trade-offs in the context of farmland abandonment: looking back into the past

Abstract

Since the second half of the last century, the landscape of European mountain areas is undergoing important changes, as a result of rural depopulation and the abandonment of traditional agricultural practices. The consequences of this process are still understudied, despite the growing concern of the possible impacts on ecosystem services and biodiversity. Most studies rely on current data and scenarios to assess the impacts of farmland abandonment on ecosystem services. Here we followed a backcasting approach using a time series of land cover maps (1960-2007) to assess changes in the condition of selected ecosystems services in a small rural mountain community in Portugal undergoing land abandonment. We determined an increasing trend in carbon storage and sequestration and sediment retention with farmland abandonment. In addition, there was a decrease in local richness of farmland plants (i.e., plant taxa with a high affinity for agricultural systems), but a relative stability of species richness for all other tested taxa. Although mountain systems are major suppliers of ecosystem services, our knowledge about the effects of land abandonment on mountain ecosystem services and their future implications for local and regional land-use planning is still scarce. This study addresses this gap by assessing post-abandonment trends in ecosystem services in a rural mountain landscape.

Keywords: biodiversity, ecosystem services, farmland abandonment, InVEST, mountain systems

4.1.Introduction

Land use change has been the main driver of biodiversity change and loss in the past century and is expected to continue in the 21st century (Leadley et al. 2010; Pereira et al. 2010). Over the past decade, considerable attention has been given to how land use changes have influenced ecosystems and the provision of ecosystem services, and ultimately human well-being (Lambin et al. 2001; Schrötter et al. 2005; Egoh et al. 2011). The conversion of natural ecosystems to human use has been mainly focused on the maximization of agricultural production (Rey Benayas and Bullock 2012). Socio-economic and demographic changes have driven land abandonment in Europe´s rural mountain landscapes since the second half of the 20th century, causing substantial land cover changes. While the majority of past studies have focused on the changes to biodiversity and ecosystem services due to intensification (Jiang et al. 2013), the abandonment of extensive agriculture and its impacts on ecosystem services has been overlooked (Tscharntk et al. 2005; Kleijn et al. 2009).

The abandonment of agricultural land led to the implementation of several measures in the EU, such as the Less Favoured Areas (LFAs-Regulation 1257/1999), implemented by the Common Agricultural Policy (CAP), promoting agricultural activity through incentives to those located in remote rural areas (Stoate et al. 2009). Yet, these efforts have been ineffective in attenuating farmland abandonment in LFA, which represent a third of Europe´s utilized agricultural area. In fact, according to recent projected scenarios, this trend is expected to continue in the future, as a consequence of ongoing rural exodus (Strijker 2005; Keenleyside and Tucker 2010).

The awareness that ecosystem services are critical for human well-being and sustain much of our economy (TEEB 2010) has resulted in the mainstreaming of ecosystem services in current biodiversity policies at global and European levels. Under the new proposal for the EU´s Common Agricultural Policy (2014-2020), the restoration and preservation of ecosystem services has been defined as a priority under the rural development pillar (European Commission 2011a). Furthermore, the EU adopted the new Biodiversity Strategy Plan for 2020, which shares goals with the global Aichi Targets of the Convention on Biological Diversity (European Commission 2011b). Specifically, the EU biodiversity target 2 refers to the restoration and maintenance of ecosystems and their services through the use of green infrastructures, and it aims at restoring at least 15% of degraded ecosystems through both EU funding and Public-Private Partnerships (European Commission 2011b). Nevertheless, for a full integration of ecosystem services into policies and land management decisions we need sound knowledge on the spatial and temporal trends of ecosystem services, so that we are able to identify synergies and trade-offs between services, providing a framing of information for both policy development and the geographical identification where targeted land use planning is needed to enhance desired services.

In recent years, researchers have developed models which facilitate assessing and quantifying the effects of policy changes and land use alteration scenarios on ecosystems, their services and human well-being (Nelson et al. 2009; Naidoo and Richetts 2006; Metzker et al. 2006, Vigerstol and Aukema 2011). Yet, backcasting changes can provide a blueprint of how the provision of ecosystem services has changed through time, which can be helpful to predict future changes based on historical trends. In this paper we quantify, spatially and temporally, the gains and losses in the capacity of a mountain landscape to supply key ecosystem services and to support biodiversity along a 47 years period, during which that landscape has suffered farmland abandonment. We focus on three ecosystem services, namely carbon sequestration and storage, sediment retention, and water yield, and on changes to biodiversity. To do so, we applied the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) software system (Natural Capital Project) to model ecosystem services, and a multi-habitat species-area relationship (SAR), developed by Pereira and Daily (2006) and parameterized to the study area by Proença and Pereira (2013) and Guilherme and Pereira (2013), to model biodiversity change.

4.2. Methods

4.2.1. Study area

Our study area, Castro Laboreiro, is a civil parish situated in the northwest of Portugal (42ºN and 8º10'W), in the Peneda-Gerês National Park (Fig. 4.1). It is a typical rural mountain community undergoing abandonment. Population numbers have decreased since the early 1960´s by 60% to today´s 540 inhabitants (INE 2011). It covers approximately 9440 hectares and the altitude gradient ranges between 300 to 1340 m, creating three distinct geographical and ecological units: the valley, the hillside, and the mountain. The valleys are characterized for their mosaic of agricultural fields, typically surrounding residential areas. Along the hillsides we can find mainly shrubland and oak forest patches. Mountain tops and plateaus are dominated by pasturing land for cattle. Oak forest patches are dominated by *Quercus robu*r and *Quercus pyrenaica,* and are one of the several habitat types listed in the Habitats Directive (92/43/CEE) occurring in the area. Temperatures vary between an average daily minimum of 0-2 Cº in the winter months and an average daily maximum of 25-28 Cº in the summer months.

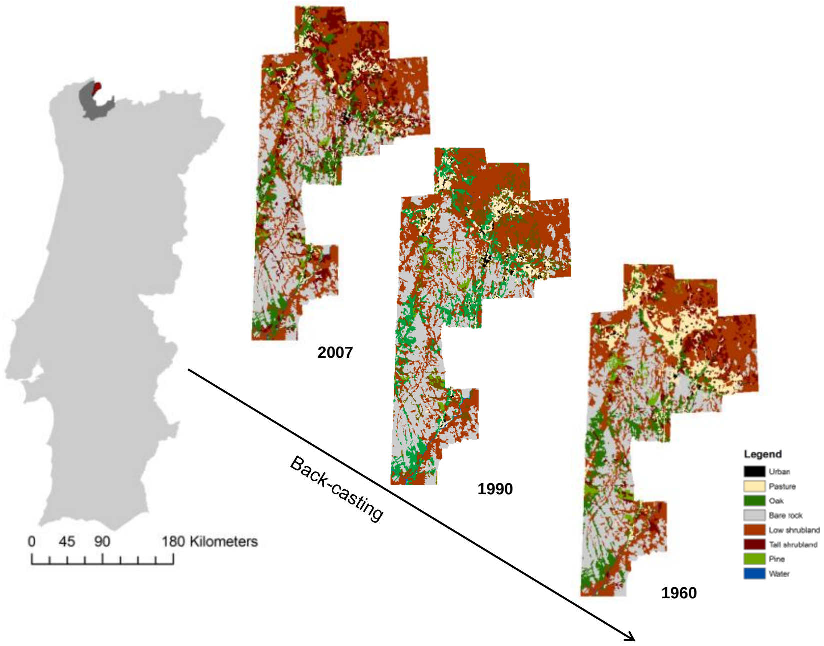

Figure 4.1 Location of the study area (dark grey: Peneda-Gerês National Park; dark red: Castro Laboreiro parish) and land cover time series (years 1960, 1990, 2007).

Castro Laboreiro provides a unique context to investigate the effects of farmland abandonment on ecosystem services. In the last 50 years it has undergone several landscape changes. Between 1960 and 2007, agricultural area has decreased by 51.5% while low and tall scrubland has become the primary land cover, jointly representing 48% of the total land cover (Rodrigues 2010).

Historically, local’s subsistence was based on agro-pastoral activities. Due to the harsh physical and environmental conditions, locals were mostly limited to farming potatoes and rye. Farmland was divided in small parcels throughout the territory, each family owing 10 to 15 parcels, with an average parcel size < 2ha (Geraldes 1996). Pastoral activity involved a nomadic seasonal migration from *brandas* (i.e., summer villages), located in the plateaus, to *inverneiras* (i.e., winter villages) found in the lower valleys. In total there are 18 inverneiras, 15 brandas and 6 permanent villages (i.e., without seasonal migration) in the Castro Laboreiro parish. The vertical movement of livestock and families from plateau settlements to valley areas occurred in December as weather conditions were most favorable in the winter villages, allowing for additional agricultural activity and pasturing land for livestock. The migration to summer villages occurred at spring time. This subsistence strategy allowed locals to exploit two distinct ecological realities creating a heterogeneous landscape. Today, this subsistence strategy is practically extinct, after social and economic changes. Today, the remaining resident farmers are in general elderly people that receive pensions and/or subsidies and practice subsistence farming.

4.2.2. Models and land use/cover maps

We applied the InVEST software (Natural Capital Project version 2.6.0) to model change in the potential supply of the following services: carbon storage and sequestration, sediment retention, and water yield, from 1960 to 2007. InVEST is a spatially explicit modelling tool that uses ecological production functions and economic valuation methods to return information on ecosystem services either in biophysical or economic terms. For biodiversity, we applied a multi-habitat SAR developed by Pereira and Daily (2006).

We used three land use/cover (LU/LC) maps corresponding to years 1960, 1990 and 2007, accessible from a previous study (Rodrigues 2010). In order to measure land use changes between each time interval, transition matrices were calculated in hectares for each LU/LC class by overlaying and intersecting two layers resulting in three transition layers (1960-1990, 1990-2007 and 1960-2007) and one trajectory layer (1960-1990 and 1990-2007). All LU/LC maps were derived from photo interpretations of aerial photographs (1960-1:8000, 1990-1:10000 and 2007-1:10000) obtained from the Portuguese Geographic Institute (IGEO). Models were run for each LU/LC map (Fig. 4.1) at a raster grid size of 30 x 30 m. Definitions for LU/LC classes are provided in Table 4.1. Details on the structure of the model and the data sources used to run the various models can be found in the Appendix (Appendix 4.1).

4.2.2.1 Biodiversity

We used a multi-habitat SAR, the countryside species-area relationship developed by Pereira & Daily (2006), to model biodiversity. This model had been previously tested and parameterized in the study area to model the diversity of plants (Proença and Pereira, 2013) and birds (Guilherme and Pereira 2013). Details on the structure of the model can be found in both studies, and full models are presented in Appendix 4.1. Briefly, the main feature of the countryside SAR is to account for the different habitats in the landscape and for the differential use of habitats by species.

Model outputs include both the number of species by groups of habitat affinity and the total number of species in the landscape. Therefore, this model has the capacity to assess changes in biodiversity due to land-use change.

Table 4.1 Land use/cover definitions for landscape interpretation (Rodrigues 2010).

LULC Class	Descriptions
Agricultural area	Crops, fallow land, pastures, meadows and fruit orchards
Low shrubland	Small shrubs: common species are *Erica* spp., *Ulex* spp. and *Halimium* spp.
Tall shrubland	Predominant species are *Cytisus* spp.
Oak forest	Forest patches of oak and other deciduous trees. Predominant species are *Quercus robur* and *Quercus pyrenaica*. Both open and closed forest
Pine forest	Forest patches of planted coniferous trees. Predominant species are *Pinus sylvestris* and *Pinus pinaster*
Rock	No vegetation cover
Water	Water surfaces, rivers and streams

Species richness estimates of plants and birds were obtained in landscape mosaics of 25 ha. A sampling grid of 500 m x 500 m was combined with LU/LC maps to derive landscape composition in 450 cells of 25ha. Landscape composition data per cell were used as an input for SAR models. Species richness was calculated for the following taxa: forest plants and birds (i.e., species showing a higher affinity for forest habitats), shrublands plants and birds (i.e., species showing a higher affinity for shrubland habitats) and agricultural plants and birds (i.e., species showing a higher affinity for shrubland habitats). The selection of these species groups was consistent with Proença and Pereira (2013) and Guilherme and Pereira (2013). In addition, it was necessary to combine tall and low shrubland in a unique category of shrublands, and to consider oak forest alone (i.e., pine forest was not included in this analysis due to its low total representation in the area).The use of a 25 ha grid enabled the visualization of trends in biodiversity change across the study area and encompassed the grain of habitat heterogeneity in the landscape.

4.2.2.2 Carbon Storage and Sequestration

We ran the InVEST Carbon Storage and Sequestration model to assess carbon stored and sequestered in the Castro Laboreiro landscape. The model aggregates C pools for each land use quantifying the amount of carbon loss or gain through time and space. To estimate carbon storage for each specific LU/LC class, we considered four carbon pools: aboveground biomass (all living plant material above the soil), belowground biomass (living root system), soil organic matter (organic component of the soil), and dead organic matter (litter and dead wood). Values for carbon pools were obtained from literature and are summarized in Appendix 4.2. The model was run using a 30 x 30 m resolution data. Each cell is assigned a LU/LC class (Appendix 4.1 and 4.2). The model sums the carbon in each of the four the pools, estimating the total carbon storage in each grid cell and in the whole landscape. We ran the model for each LU/LC map (1960, 1990 and 2007). The net change in carbon storage from 1960 to 2007 was determined by subtracting the carbon stored in 1960 in each grid cell from that stored in 2007. Output values are represented in Mg of carbon per grid cell. Positive values indicate the net sequestration of carbon, whereas negative values indicate a net loss of carbon.

4.2.2.3 Sediment Retention

We applied the InVEST Avoided Reservoir Sedimentation model to calculate the average annual soil loss from each parcel of land, sediment exported as well as the ability of each parcel to retain sediment. The capacity of a land cover to retain sediment depends on geomorphology, climate, vegetation and management practices. The model uses the Universal Soil Loss Equation (USLE) at the pixel level, where the rate of soil erosion is dependent on land cover/use, rainfall erosivity, soil erodibility and the length of slope. Potential soil conservation was mapped as the inverse of the USLE outputs (Nelson et al. 2009). Data sets and sources used are summarized in the supplementary material (Appendices 4.1 and 4.3). Model outputs are interpreted at the sub-watershed level. Output values are represented in tons/ha.

4.2.2.4 Water Yield

We used the InVEST model to assess the influence of land use/cover on water yield between 1960 and 2007 in the Castro Laboreiro river watershed. Specifically, we assessed how agriculture conversion to abandonment impacted water yield through time. Water yield in this study represents the precipitation that did not evaporate from both ground or water surfaces and did not transpire from plant surfaces. To run the model, the main biophysical parameters included climatic, geomorphologic information and land use characteristics. The data used to run the model are summarized in the supplementary material (Appendices 4.1 and 4.3). Interpretations of the results were done at the sub-watershed level.

4.2.2.5 Analysis of Ecosystem services: Trade-offs

To evaluate the spatial correspondence of the various ecosystem services and biodiversity between 1960 and 2007 we first overlapped a 500 x 500m grid to each of the service maps. We used the Spatial Analyst raster calculator tool of ArcGIS to calculate the mean value for each ecosystem service per cell and subtracted the values between the two time periods. To spatially analyze the trade-offs we reclassified the values, attributing a value of 0 to those cells that had remained constant from 1960 to 2007, 1 to those where there was a gain, and -1 to those that suffered a decrease or loss for each ecosystem service and biodiversity. We then proceeded to summing the various ecosystem services and biodiversity, resulting in a spatially explicit evaluation of the total gains and losses per grid cell. The transitional values ranged from 4 to -4, with 4 being the maximum number of services gained per grid cell, and -4 the highest loss of services.

4.3. Results

4.3.1. Biodiversity

Land use changes in Castro Laboreiro have mainly affected plant species associated with agricultural systems, with an average loss of four species (-3.95) per 25 ha grid cell (Appendix 4.4). We further analyzed a sub-sample of 212 cells where we considered only cells in which the sum of the three habitats with highest species affinity (agriculture, oak and scrubland) was >15 ha. Here, the richness of all other taxa remained relatively stable with minor average changes, positive or negative, of less than one species gained or lost per 25 ha habitat mosaic (Fig. 4.2 and 4.3).

4.3 2. Carbon Storage and Sequestration

Carbon storage was calculated for each land cover class, for aboveground and belowground biomass, soil and dead organic matter for each time period (Appendices 4.5-7). We determined that forest systems had the highest capacity for carbon storage, followed by scrubland. For these particular land cover classes, carbon storage mostly occurs in the soil and aboveground biomass, with low values for dead organic matter. Agriculture had the second lowest carbon storage values, after urban areas. Scrubland and forest contributed the most to the whole landscape carbon storage. In 2007, both tall and low scrubland contributed 61% of the total carbon landscape, followed by forest systems, pine and oak, with 35%.

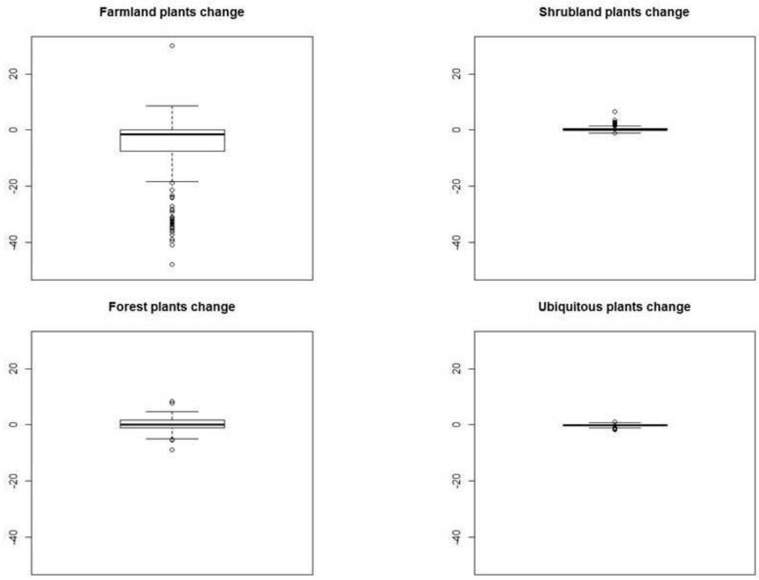

Figure 4.2 Estimated change in the richness of plant species groups (1960-2007) per 25 ha (n = 212 cells). Note that only cells in which the sum of agricultural area + scrubland + oak forest > 15 ha were used.

The total carbon stocks in the landscapes of 1960, 1990 and 2007 were respectively 8.4 x 10^5 Mg, 8.5 x 10^5 Mg, and 8.6 x 10^5 Mg (Fig. 4.4a-c).

Studying the amount of carbon sequestered locally helps to identify policies which maximize the lands overall potential. Past analysis on land cover trajectory showed an increase in oak and tall scrubland cover, between the years 1960 to 2007, from 1021 ha to 1059 ha and from 868 ha to 1513 ha, respectively. Although the increase for oak cover is minimal, the region has suffered a substantial decrease in agricultural area, from 1212 ha to 560 ha.

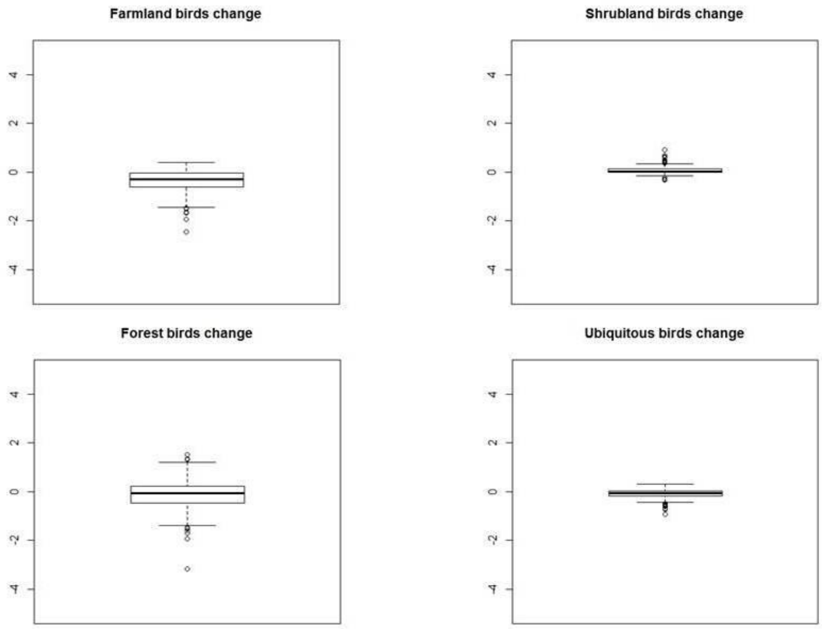

Figure 4.3 Estimated change in the richness of bird species groups (1960-2007) per 25 ha (n=212 cells). Note only cells in which the sum of agricultural area + scrubland +oak forest >15 ha were used.

Our results show that, throughout the landscape, these changes translated into both carbon gains and losses from 1960 to 2007 (Fig. 4.4d). Total net carbon sequestration from 1960 to 2007 was estimated to be 11269 Mg. In Figure 4.4(d), large negative values are mostly areas where there has been conversion from land cover types with high carbon storage potential to low carbon storage (e.g. pine to low scrubland). On the other hand, higher positive values represent areas of high carbon sequestration, corresponding mainly to areas in which there has been minimal land conversion along the 47 year span or then increased restoration of forest systems and scrubland. Our results also show that there is a fairly high representation of areas with no net carbon sequestration; these are mainly associated to bare rock and low shrubland.

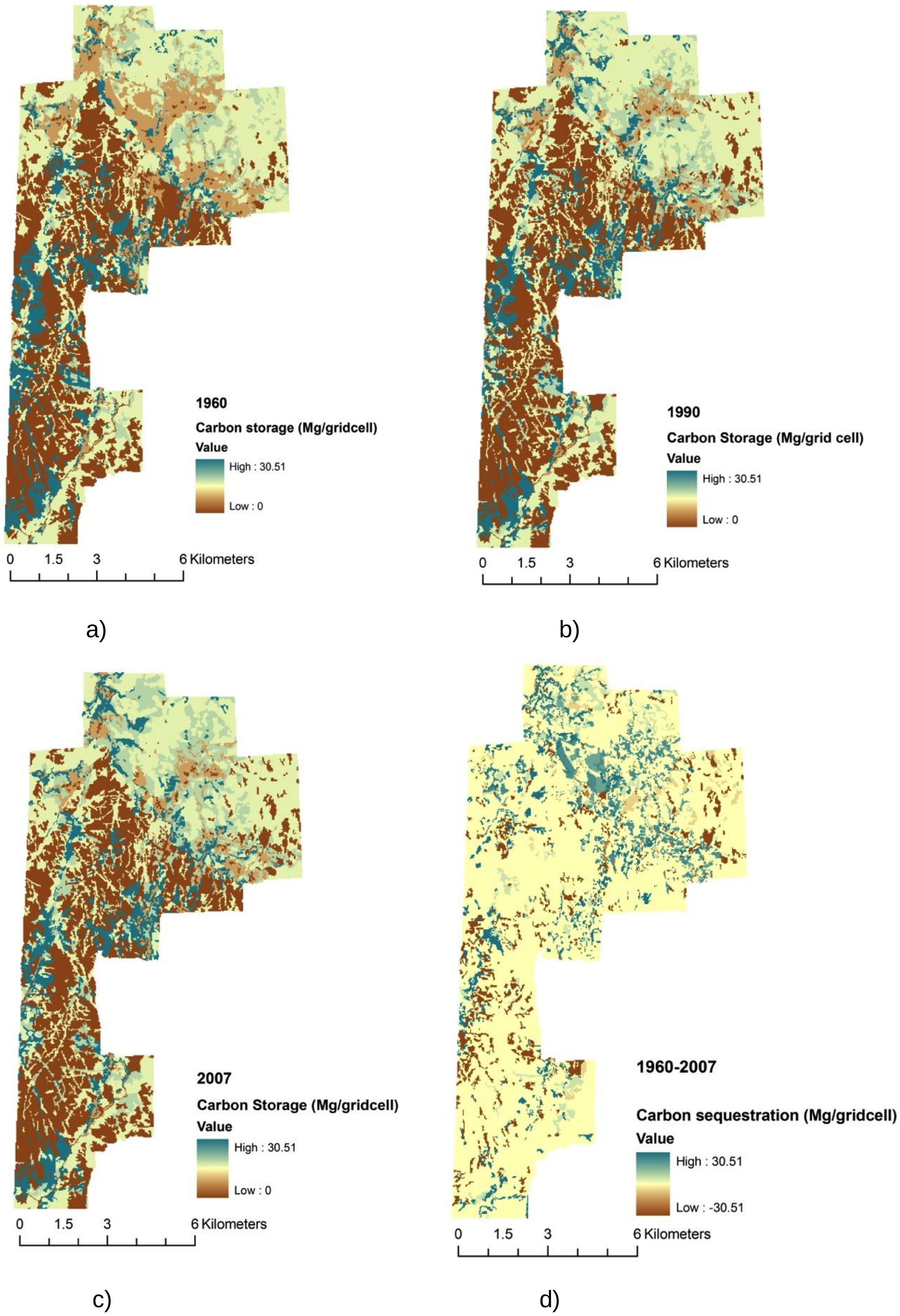

Figure 4.4 (a-c) estimates of carbon storage for each land use for the following years; 1960, 1990 and 2007; d) total sequestered carbon between 1960 and 2007.

4.3.3. Sediment Retention

Using the USLE we calculated the potential soil loss for Castro Laboreiro. Our study estimated a 45% decrease of potential soil loss between 1960 (5.28 ton/ha) and 2007 (2.40 ton/ha). We also determined soil conservation using the inverse values of the estimated soil loss. Through the mapping of the values we are able to identify those areas of high and low soil conservation (Fig. 4.5a-c). Areas of low soil conservation are mainly associated to regions of rock cover, in contrast to areas with high soil conservation, which are mainly forest areas. The north of the study area shows high soil conservation in comparison to the rest of the landscape. Generally, we see a positive trend for soil conservation along the years, specifically the south and eastern regions. From our analysis, we are thus able to identify where are the sediment retention areas. Our results also highlight a decrease of 57% on sediment exported between 1960 (5508 tons/ha) and 2007 (2798 tons/ha), which is likely associated to the decrease in land use intensity along the years.

4.3.4. Water Yield

Our results show that water yield increased from 1960 to 2007 (Fig. 4.6a-c). We determined that the changes in mean annual water yield fluctuated across the nine sub-watersheds. We obtained increases ranging from 43 m^3/ha (4%) to 2 m^3/ha (2%), and a decrease of 10 m^3/ha (1%) in one sub-watershed. The most significant increases occurred in both the eastern and western sub-basins. Changes in the water yield were less evident in the middle and southern sub-basins.

In both the eastern and western parts of the catchment there were significant pasture decreases and tall scrubland encroachment, whereas in the south, land use trajectory suffered less obvious changes (Figure 4.6). Decreased agricultural land in both the eastern through to the western parts of the catchment has led to increases in water yield in the lower part of the catchment.

The results of the model also identified a decrease in evapotranspiration in those sub-watersheds which had from 1960 to 2007 revealed an increase in terms of water yield. These ranges go from -2.3 mm (0.5 %) to -43 mm (18.2%). Only one sub-watershed (2) underwent a 10.6 mm (3.9%) increase in evapotranspiration, the same sub-watershed that suffered a decrease in water yield. By comparing the land cover maps with the spatial outputs from the water yield models, we conjecture a potential relation for areas of high vegetation cover (i.e. forests) and lower water yield. In other words, high forest cover results in decreased water supply at the watershed level.

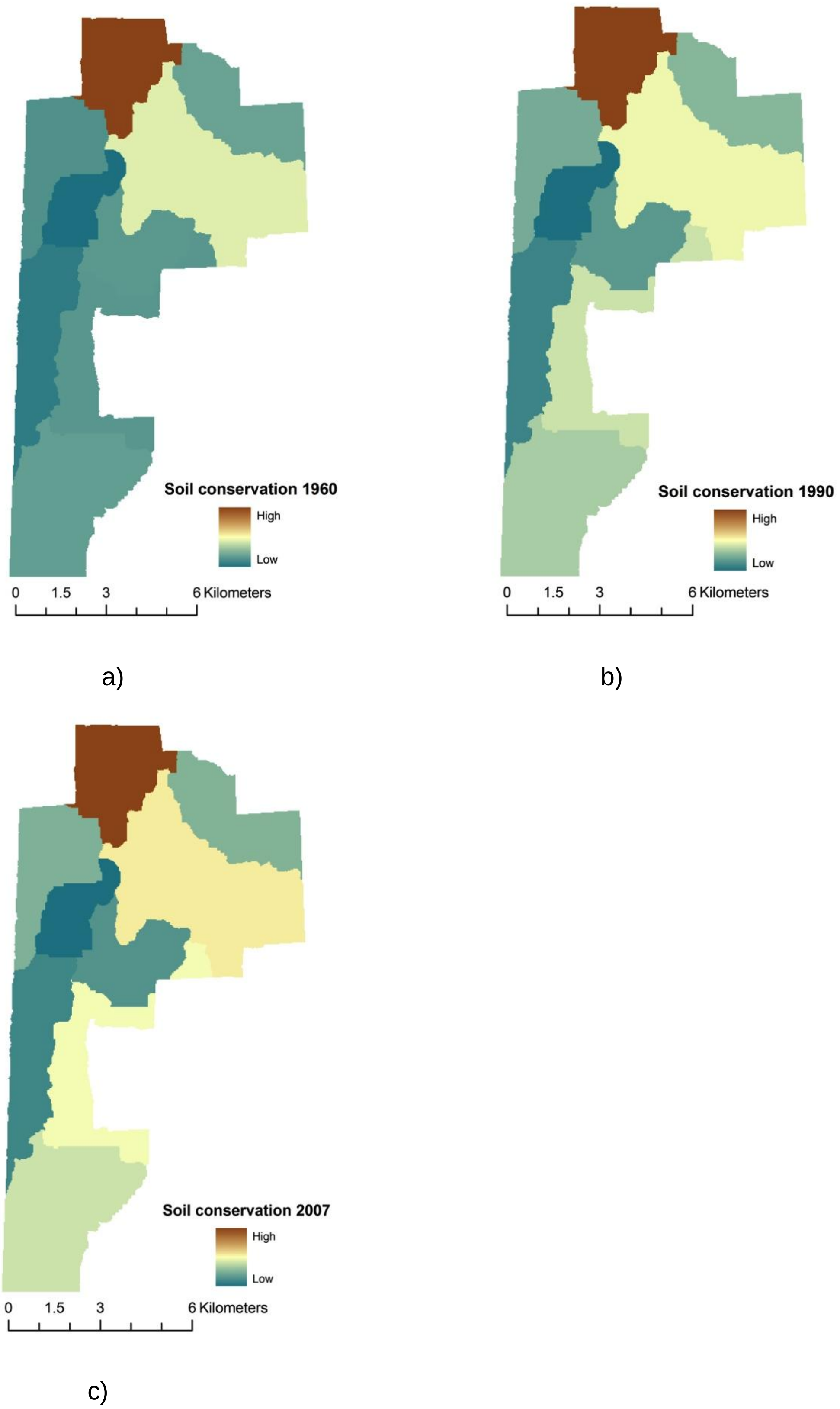

Figure 4.5 Soil conservation in (a) 1960, (b) 1990, and (c) 2007, at sub-watershed level.

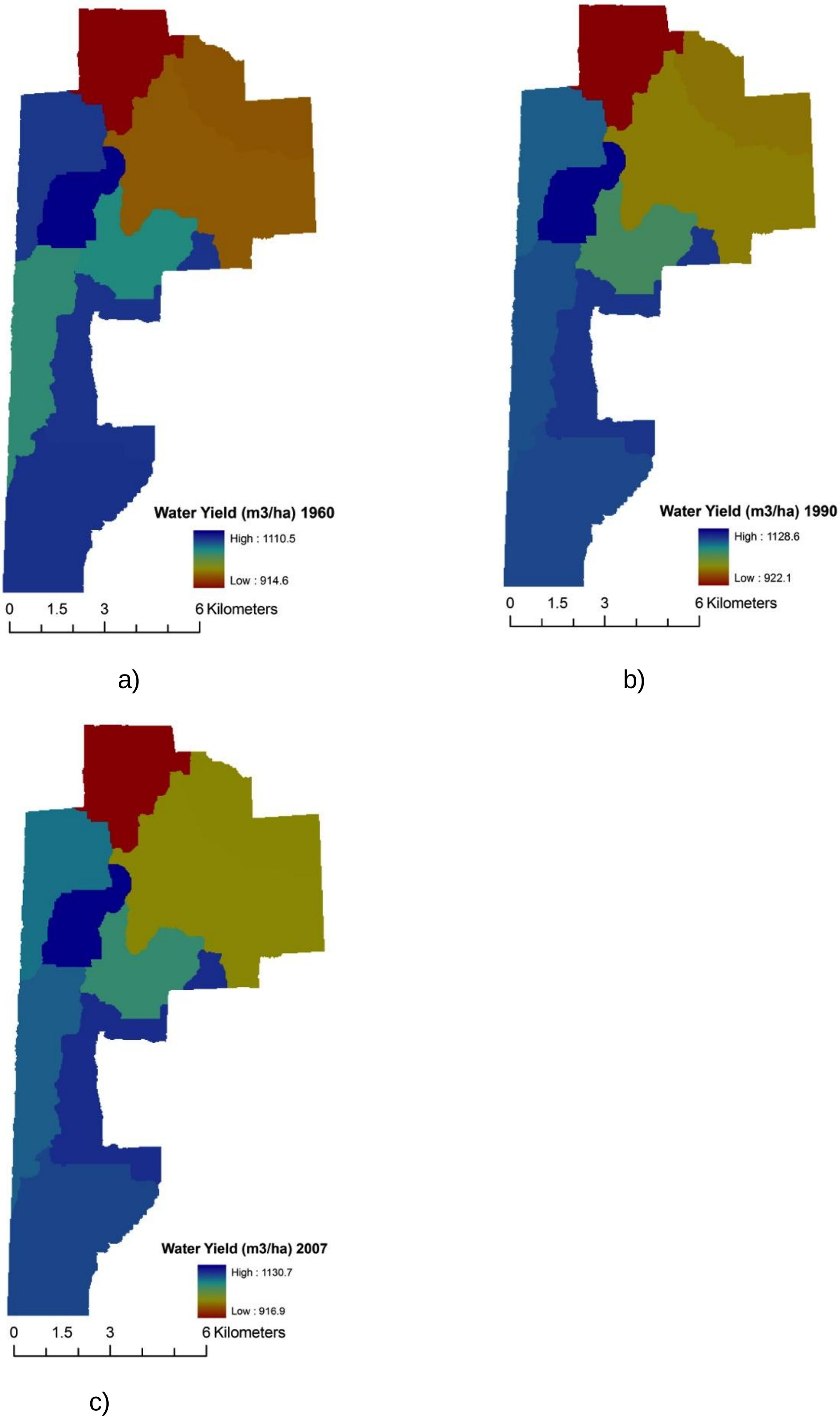

Figure 4.6 (a-c) Model estimates for water yield volumes (m^3) in the Castro Laboreiro river watershed for years 1960, 1990 and 2007.

4.3.5. Trade-off analysis for biodiversity and ecosystem services

We finally assessed the spatial gains and losses of ecosystem services and biodiversity at the landscape level. According to our results there are specific regions in which there are increases in several of the focal ecosystem services with on-going land abandonment (Fig. 4.7). The visualization of these gains is important in conservation efforts. We can speculate that those areas that did not experience gains or losses did not undergo land use change between the period of 1960 and 2007. Very few areas were indicative of decreases for 3 or all 4 ecosystem services. The analysis also revealed that those areas of higher gains were generally distributed continuously, permitting the identification of hotspots (Fig. 4.7). Although statistically we did not test for the association of land cover and ecosystem service gains and losses, we did carry out a simple overlay of the land cover maps and found that the majority of these areas were association to forested areas.

4.4. Discussion

Most studies rely on current data and exploratory future scenarios to assess the impacts of land use change on ecosystem services (Plieninger et al. 2013). In the present study, we followed a back-casting approach using a time series of land cover maps (1960-2007) to assess changes in estimated values for various ecosystem services and biodiversity as well as the spatial distribution of these services in face of land abandonment. Through the use of general models we estimated alterations to the variables associated to ecosystem services. These alterations include a positive effect on most of the focal ecosystem services. For biodiversity, we estimated an average decrease in local richness of farmland plants (i.e., plant taxa with a higher affinity for agricultural systems), but a relative stability of the richness of all other tested taxa, in agreement with other studies (Lomba et al. 2012; Dornelas et al. 2014).

Being able to identify and measure changes in land cover provides a better understanding of how ecosystem services fluctuate under different levels of anthropogenic interference. Our results suggest that decreasing human influence positively impacts the trend of various ecosystem services, linked to the restoration or expansion of those ecosystems suppressed by past agricultural activity (Navarro and

Pereira 2012). We determined that the majority of the changes for the various ecosystem services and biodiversity occurred between 1960 and 1990, coinciding with the period of greatest land cover change (Rodrigues 2010). However, we found that some indicators such as total carbon stock did not drastically increase in relation to a decrease in land use intensity along the years, contrary to past findings (Grimaldi et al. 2014; Kuemmerle et al. 2011). We relate these results to the increased fire occurrence in the region and the subsequent suppression of forest expansion (Moreira et al. 2001;

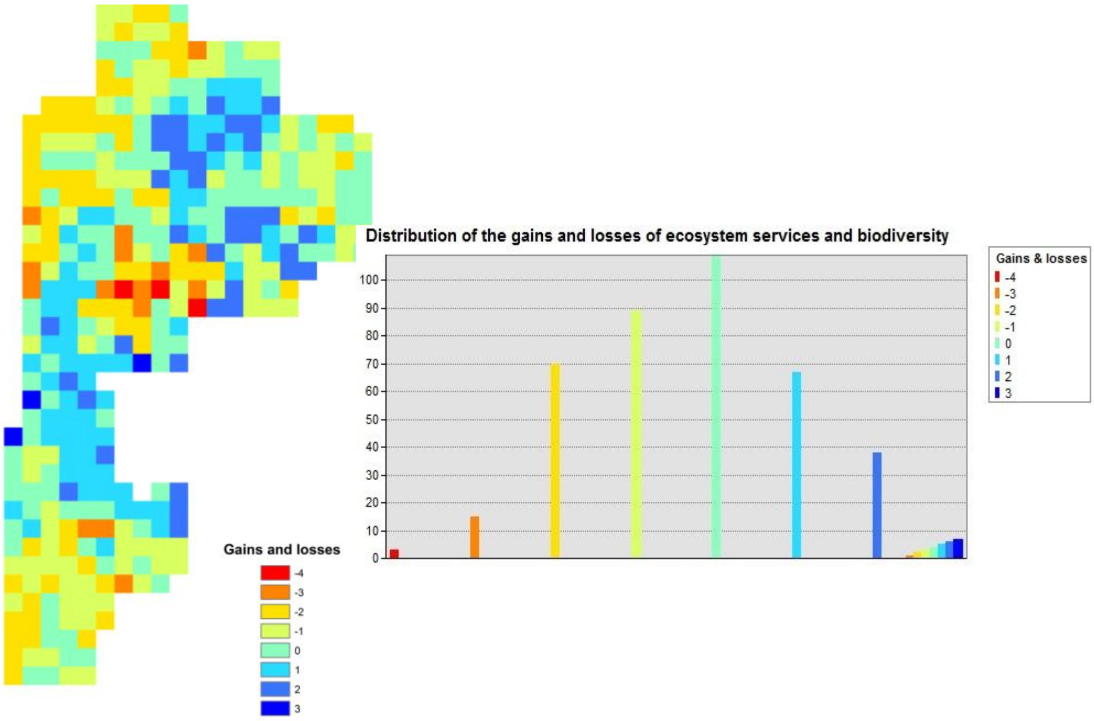

Figure 4.7 Trade-off analysis for biodiversity, carbon storage, sediment retention and water yield in Castro Laboreiro, from 1960 to 2007. A gradient of colors from red to dark blue is used to depict the losses, no changes and gains for the focal ecosystem services and biodiversity (red= highest losses, dark blue=highest gains, and light blue=no change). The histogram shows how the values are statistically distributed.

Rodrigues 2010; Torres-Manso et al. 2014). Nonetheless, we can envision an on-going forest expansion in the future based on past and future drivers, delineated in Beilin et al. (2014). This expansion will naturally lead to a higher potential carbon sequestration, playing a role in climate regulation and other relevant services.

The restoration or enhancement of forest cover is fundamental in underpinning other essential processes and services (Vidal-Legaz et al. 2013;Qiu and Turner 2013; Raudsepp-Hearne et al. 2010). These include high infiltration rate, reducing peak flows and controlling natural hazards and river regimes and improving downstream water quality (Stoate et al. 2001; Körner 2002; Maes et al. 2009; Santos de Lima et al. 2014). According to a recent study carried out in Castro Laboreiro, there are also ecological benefits in the expansion of native broadleaved forests (Proença et al. 2010), as native forests have higher resistance and resilience after fire occurrence, which contributes to a greater stability of landscapes and therefore of forest processes and ecosystem services. Nonetheless, we cannot omit the fact that the occurrence of fire in mountain regions has increased, which has been linked to an increase of fuel biomass after land abandonment (Moreira et al. 2001; Torres-Manso et al. 2014). The protection of forests, especially old growth forests, is important, supplying higher levels of biodiversity, water and nutrient services in comparison to young forests (Ferraz et al. 2014; Onaindia et al. 2013). The awareness of the importance of old growth forests has resulted in its incorporation in the EU Biodiversity Strategy through the protection of wilderness (European Commission 2011a). This re-enforces the premise that restoration or expansion of natural habitats ensures and promotes the sustainability of ecological processes while contributing to the new biodiversity strategy.

Using the maps as reference we were able to determine the spatial gains and losses of ecosystem services and biodiversity. Areas of highest gains are mainly those regions which have not suffered any losses due to on-going land abandonment. These are mainly land cover types associated to forest or areas undergoing decreases in land use intensity (Vidal-Legaz et al. 2013; Jiang et al. 2013). By examining the spatial gains and losses of total ecosystem services from 1960 to 2007 (see Fig. 4.7), we were able to identify hotspots. These hotspots are distributed throughout the landscape, however there is some connectivity between areas of high gains. We speculate that those areas which have suffered decreases in service provision may be linked to areas of active farming or that have recently been burned.

Due to the expected on-going abandonment of rural mountains, alternative land management strategies are now being examined to identify a win-win opportunity for multiple ecosystem services and biodiversity. A recent study assessed potential future land management strategies throughout Europe, and their possible impacts on several ecosystem services (see Navarro and Pereira 2012). Overall, their qualitative assessment revealed that in rewilded areas a wider range of services are provided in comparison to all other management strategies presented. The natural consequence of land abandonment is the restoration of the natural vegetation through rewilding, and this is considered a possible potential strategy for Europe´s rural mountain landscapes.

Though there are numerous benefits connected to land abandonment, it is acknowledged that it can be prejudicial to both species and landscape diversity (Bielsa et al. 2005; Russo 2006; Beilin et al. 2014; Lomba et al. 2014; Zakkak et al. 2014). However, these consequences are dependent on the species being evaluated and the composition of the landscape before abandonment has taken place. Russo (2006) and most recently Navarro and Pereira (2012) identified a number of winner and loser species at the European scale, benefiting or losing from rewilding and abandonment of farmland. In particular, the later study acknowledged a high potential of habitat for biodiversity in rewilded areas.

Our findings confirmed that species richness for flora with affinity to extensive agriculture decreased (Lomba et al. 2012), but the remaining groups exhibited little changes to species richness (see Fig. 4.2 and 4.3). We speculate that the impacts of land use changes in plant and bird communities were buffered by the ability of species to use several habitats available in the landscape (Proença and Pereira 2013). While the principle that we needed to preserve agro-biodiversity has led to the endorsement of traditional agricultural practices through agri-environmental schemes, this has not been effective in halting rural exodus and farmland abandonment in rural mountain systems (Kleijn and Sutherland 2003; MacDonald et al. 2000). Additionally, a recent study highlighted that intensifying land use in a rural mountain setting would not reverse the demographic effects and would consequently lead to a negative impact, on both water supply and aesthetic services (Vidal-Legaz et al. 2013). Another solution could be one that supports Phalan et al.’s (2011) land sparing approach, through the separation of biodiversity and agriculture but with a particular emphasis on ecosystem services. For example, in the present study we have identified those areas with gains and those with losses (see Fig. 4.7). In this case those with losses could potentially be areas of agricultural use, promoting not only agro-biodiversity but the conservation of various ecosystem services endorsed due to decreases in land use intensity.

By using a back-casting approach, we obtained a glimpse of how particular services have responded to land abandonment. Although the model outputs do not deliver precise values, they are estimates of the alterations to the values of the various ecosystem services, enabling a visualization of the potential impacts of land abandonment in a rural mountain. Even so, there is a considerable amount of uncertainties associated to the data gathered from the literature to run the various models. The data were not locally specific, potentially skewing our results. The study also did not consider data on food production due to lack of available data sources. This information could have provided a better insight on where we presently have gains for the service and where there have been losses. This visualization is important for landscape planning and more specifically in identifying suitable management strategies.

The protection of rural mountains has become a key target in conservation efforts. These are fragile systems, which are subject to anthropogenic and natural drivers of change, recovering slowly or not at all from disturbances, due to their characteristic sloping terrain and thin soils (Körner and et al. 2005). Disturbances to mountain ecosystems consequently decrease ecosystem function, impacting lowland areas and, overall, human well-being (Körner et al. 2005). These regions are not only hotspots of ecosystem services but also important for biodiversity (Körner 2002; Harrison et al. 2010; Maes et al. 2012). In many cases, the benefits are geographically dispersed, which means that the provisioning area isn´t the only region benefiting from the services, but also those communities downstream (Serna-Chavez et al. 2014). For example, soil erosion control in mountain systems has become an indispensable service considered in natural management decisions due to the multiple cascading negative environmental effects (Vanacker et al. 2014). By protecting these fragile systems, we are ultimately contributing to the sustainability of ecosystem processes and functions while promoting multi-functionality (Reyers et al. 2012; Schindler et al. 2014) and potentially generating an economy that is “nature-friendly”, based on ecosystem services and biodiversity (Ervin et al. 2012; Navarro and Pereira 2012).

4.5. References

Beilin, R., Lindborg, R., Stenseke, M., Pereira, H. M., Llausàs, A., Slätmo, E., … Queiroz, C. (2014). Analysing how drivers of agricultural land abandonment affect biodiversity and cultural landscapes using case studies from Scandinavia, Iberia and Oceania. *Land Use Policy*, *36*(0), 60–72. doi:10.1016/j.landusepol.2013.07.003

Bielsa, I., Pons, X., & Bunce, B. (2005). Agricultural Abandonment in the North Eastern Iberian Peninsula: The Use of Basic Landscape Metrics to Support Planning. *Journal of Environmental Planning and Management*, *48*(1), 85–102. doi:10.1080/0964056042000308166

Dornelas, M., Gotelli, N. J., McGill, B., Shimadzy, H., Moyes, F., Sievers, C., Magurran, A. 2014.Assemblage Time Series Reveal Biodiversity Change but Not Systemic Loss. Science 344, 296.

Egoh, B. N., Reyers, B., Rouget, M., Richardson, D. M. (2011). Identifying priority areas for ecosystem service management in South African grasslands. *Journal of Environmental Management*, 1-9.

Ervin, D., Brown, D., Chang, H., Dujon, V., Granek, E., Shandas, V., & Yeakley, A. (2012). Growing Cities Depend on Ecosystem Services. *Solutions*, *2*(6), 74–86.

European Commission, 2011a. Regional policy contributing to sustainable growth in Europe 2020. COM(2011)17 final. Brussels.

European Commission, 2011b. Our life insurance, our natural capital: an EU biodiversity strategy to 2020. Brussels, 2011-05-03. COM(2011) 244.

Ferraz, S. B., Ferraz, K. P. M. B., Cassiano, C., Brancalion, P., Luz, D. A., Azevedo, T., … Metzger, J. (2014). How good are tropical forest patches for ecosystem services provisioning? *Landscape Ecology*, *29*(2), 187–200. doi:10.1007/s10980-014-9988-z

Geraldes A. D. (1996) Brandas e Inverneiras. Particularidades do sistema agro-pastoril castrejo. Cadernos Juríz Xurés. Instituto da Conservação da Natureza e Parque Nacional da Peneda Gerês.

Grimaldi, M., Oszwald, J., Dolédec, S., Hurtado, M. delPilar, de Souza Miranda, I., Arnauld de Sartre, X., … Lavelle, P. (2014). Ecosystem services of regulation and support in Amazonian pioneer fronts: searching for landscape drivers. *Landscape Ecology*, *29*(2), 311–328. doi:10.1007/s10980-013-9981-y

Guilherme, J. L., & Miguel Pereira, H. (2013). Adaptation of Bird Communities to Farmland Abandonment in a Mountain Landscape. *PLoS ONE*, *8*(9), e73619. doi:10.1371/journal.pone.0073619

Harrison, P., Vandewalle, M., Sykes, M., Berry, P., Bugter, R., Bello, F., … Zobel, M. (2010). Identifying and prioritising services in European terrestrial and freshwater ecosystems. *Biodiversity and Conservation*, *19*(10), 2791–2821. doi:10.1007/s10531-010-9789-x

INE - Instituto Nacional de Estatística (2011). Recenseamento da população e habitação. INE. Lisboa,Portugal.

Jiang, M., Bullock, J. M., & Hooftman, D. A. P. (2013a). Mapping ecosystem service and biodiversity changes over 70 years in a rural English county. *Journal of Applied Ecology*, *50*(4), 841–850. doi:10.1111/1365-2664.12093

Keenleyside C. & Tucker, G. (2010). Farmland Abandonment in the EU: an assessment of trends and prospects. London: WWF and IEEP. 97 pp.

Kleijn, D., Kohler, F., Báldi, A., Batáry, P., Concepción, E. ., Clough, Y., … Verhulst, J. (2009). On the relationship between farmland biodiversity and land-use intensity in Europe. *Proceedings of the Royal Society B: Biological Sciences*, *276*(1658), 903–909.

Kleijn, D., & Sutherland, W. J. (2003). How effective are European agri-environment schemes in conserving and promoting biodiversity? *Journal of Applied Ecology*, *40*(6), 947–969. doi:10.1111/j.1365-2664.2003.00868.x

Körner, C. and M. Ohsawa (coordinating lead authors), et al. 2005. Mountain Systems. Chapter 24. In: R. Hassan, R. Scholes, and N. Ash, (eds.). Ecosystems and human well-being: current state and trends, Volume 1. Washington, DC: Island Press. p. 681-716.

Körner, C. (2002). “Mountain biodiversity, its causes and function: an overview”. In: Korner, C. & Spehn, E. (eds), *Mountain biodiversity: a global assessment*. Parthenon, New York, pp. 3-20.

Kuemmerle, T., Olofsson, P., Chaskovskyy, O., Baumann, M., Ostapowicz, K., Woodcock, C. E…..Radeloff, V.,C. (2011). Post-Soviet farmland abandonment, forest recovery, and carbon sequestration in western Ukraine. *Global Change Biology*, *17*(3), 1335–1349. doi:10.1111/j.1365-2486.2010.02333.x

Lambin, E. F., & Meyfroidt, P. (2011). Global land use change, economic globalization, and the looming land scarcity. *Proceedings of the National Academy of Sciences*, *108*(9), 3465–3472.

Leadley, P., Pereira, H.M., Alkemade, R., Fernandez-Manjarres, J.F., Proenca, V., Scharlemann, J.P.W., Walpole, M.J. (2010) Biodiversity Scenarios: Projections of 21st century change in biodiversity and associated ecosystem services. Secretariat of the Convention on Biological Diversity, Montreal. Technical Series no. 50, 132 pages.

Lomba, A., Gonçalves, J., Moreira, F., Honrado, J. 2012. Simulating long-term effects of abandonment on plant diversity in Mediterranean mountain farmland, Plant Biosystems - An International Journal Dealing with all Aspects of Plant Biology: Official Journal of the Societa Botanica Italiana, DOI:10.1080/11263504.2012.716794

Lomba, A., Gerra, c., Alonso, J., Honrado, J., Jongmanb, R., McCracken, D. (2014). Mapping and monitoring High Nature Value farmlands: Challenges in European landscapes. Journal of Environmental Management, 143:140-150.

MacDonald, D., Crabtree, J. ., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., … Gibon, A. (2000). Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. *Journal of Environmental Management*, *59*(1), 47–69. doi:10.1006/jema.1999.0335

Maes, J., Paracchini, M. L., Zulian, G., Dunbar, M. B., & Alkemade, R. (2012). Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. *Biological Conservation*, *155*(0), 1–12. doi:10.1016/j.biocon.2012.06.016

Maes, W. H., Heuvelmans, G., & Muys, B. (2009). Assessment of Land Use Impact on Water-Related Ecosystem Services Capturing the Integrated Terrestrial−Aquatic System. *Environmental Science & Technology*, *43*(19), 7324–7330. doi:10.1021/es900613w

Metzger, M. J., Rounsevell, M. D. A., Acosta-Michlik, L., Leemans, R., & Schröter, D. (2006). The vulnerability of ecosystem services to land use change. *Scenario-Based Studies of Future Land Use in Europe*, *114*(1), 69–85. doi:10.1016/j.agee.2005.11.025

Moreira, F., Rego, F., & Ferreira, P. (2001). Temporal (1958–1995) pattern of change in a cultural landscape of northwestern Portugal: implications for fire occurrence. *Landscape Ecology*, *16*(6), 557–567. doi:10.1023/A:1013130528470

Naidoo, R., & Ricketts, T. H. (2006). Mapping the Economic Costs and Benefits of Conservation. *PLoS Biol*, *4*(11), e360. doi:10.1371/journal.pbio.0040360

Navarro, L., & Pereira, H. (2012). Rewilding Abandoned Landscapes in Europe. *Ecosystems*, *15*(6), 900–912. doi:10.1007/s10021-012-9558-7

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, Dr., … Shaw, Mr. (2009). Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment*, *7*(1), 4–11. doi:10.1890/080023

Onaindia, M., Fernández, D. M., Madariaga, I., Rodríguez-Loinaz, G. 2013. Co-benefits and trade-offs between biodiversity, the carbon storage, and water flow regulations. Forest Ecology Management, 289:1-9.

Pereira, H. M., Leadley, P. W., Proença, V., Alkemade, R., Scharlemann, J. P. W., Fernandez-Manjarrés, J. F., … Walpole, M. (2010). Scenarios for Global

Biodiversity in the 21st Century. *Science*, *330*(6010), 1496–1501.

Pereira, H.M., Daily, G. C. (2006). Modeling Biodiversity Dynamics in Countryside *Landscapes Ecology*, *87:*1877-1885.

Plieninger, T., Bieling, C., Ohnesorge, B., Schaich, H., Schleyer, C., & Wolff, F. (2013). Exploring Futures of Ecosystem Services in Cultural Landscapes through Participatory Scenario Development in the Swabian Alb, Germany. *Ecology and Society*, *18*(3). doi:10.5751/ES-05802-180339

Proença, V., & Pereira, H. M. (2013). Species–area models to assess biodiversity change in multi-habitat landscapes: The importance of species habitat affinity. *Basic and Applied Ecology*, *14*(2), 102–114. doi:10.1016/j.baae.2012.10.010

Proença, V., Pereira, H. M., Vicente, L. (2010). Resistance to wildfire and early regeneration in natural broadleaved forest and pine plantation. *Acta Oecologica, 36*:626-633.

Qiu, J., & Turner, M. G. (2013). Spatial interactions among ecosystem services in an urbanizing agricultural watershed. *Proceedings of the National Academy of Sciences*. Retrieved from http://www.pnas.org/content/early/2013/06/27/1310539110.abstract

Raudsepp-Hearne, C., Peterson, G. D., Bennett, E. M. 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. PNAS, 1-6.

Rey Benayas, J., & Bullock, J. (2012). Restoration of Biodiversity and Ecosystem Services on Agricultural Land. *Ecosystems*, *15*(6), 883–899. doi:10.1007/s10021-012-9552-0

Reyers, B., O’Farrell, P., Nel, J., & Wilson, K. (2012). Expanding the conservation toolbox: conservation planning of multifunctional landscapes. *Landscape Ecology*, *27*(8), 1121–1134. doi:10.1007/s10980-012-9761-0

Rodrigues, P. (2010). Landscape changes in Castro Laboreiro: from farmland abandonment to forest regeneration. Masters Thesis, University of Lisbon.

Russo, D. (2006). Effects of land abandonment on animal species in Europe: conservation and management implications. Integrated assessment of vulnerable ecosystems under global change in the EU. Project report. 52 pp.

Santos de Lima L., Coe, M.T., Soares-Filho, B.S., Cuadra, S.V., Pinto Dias, L.C., Costa, M.H,. Santos de Lima, L., Rodrigues, H.O .2014. Feedbacks between deforestation, climate, and hydrology in the Southwestern Amazon: implications for the provision of ecosystem services. Landscape Ecol. doi:10.1007/s10980-013-9962-1

Schindler, S., Sebesvari, Z., Damm, C., Euller, K., Mauerhofer, V., Schneidergruber, A., … Wrbka, T. (2014). Multifunctionality of floodplain landscapes: relating management options to ecosystem services. *Landscape Ecology*, *29*(2), 229–244. doi:10.1007/s10980-014-9989-y

Schrötter, D., Cramer, W., Leemans, R., Prentice, I. C., Araújo, M. B.,Arnell, N. W., Bondeau, A., Bugmann, H., Carter, T. R., Gracia, C., Vega-Leinert, A. C., Erhard, M., Ewert, F., Glendining, M., House, J. I., Kankaanpaa, S., Klein, R. J. T et al. (2005). Ecosystem Service Supply and Vulnerability to Global Change in Europe. *Science, 310*, 1333-1337.

Serna-Chavez, H. M., Schulp, C. J. E., van Bodegom, P. M., Bouten, W., Verburg, P. H., & Davidson, M. D. (2014). A quantitative framework for assessing spatial flows of ecosystem services. *Ecological Indicators*, *39*(0), 24–33. doi:10.1016/j.ecolind.2013.11.024

Stoate, C., Báldi, A., Beja, P., Boatman, N. D., Herzon, I., van Doorn, A., … Ramwell, C. (2009). Ecological impacts of early 21st century agricultural change in Europe – A review. *Journal of Environmental Management*, *91*(1), 22–46. doi:10.1016/j.jenvman.2009.07.005

Strijker, D. (2005). Marginal lands in Europe–causes of decline. *Basic Appl Ecol, 6*:99–106.

TEEB. 2010. The economics of ecosystems and biodiversity: mainstreaming the economics of nature: a synthesis of the approach, conclusions and recommendations of TEEB. 39 pp.

Terres, J. M., Nisini, L. 2012. Assessing the risk of farmland abandonment in the EU. European Commission. 138 pp. EUR – Scientific and Technical Research series – ISSN 1831-9424

Torres-Manso, F., Fernandes, P., Pinto, R., Botelho, H., Monzon, A. 2014.Regional livestock grazing, human demography and fire incidence in the Portuguese landscape. Forest Systems, 23(1): 15-21.

Tscharntke, T., Klein, A. M., Kruess, A., Steffan-Dewenter, I., & Thies, C. (2005). Landscape perspectives on agricultural intensification and biodiversity – ecosystem service management. *Ecology Letters, 8*: 857–874. doi: 10.1111/j.1461-0248.2005.00782.x

Vanacker, V., Bellin, N., Molina, A., & Kubik, P. (2014). Erosion regulation as a function of human disturbances to vegetation cover: a conceptual model. *Landscape Ecology*, *29*(2), 293–309. doi:10.1007/s10980-013-9956-z

Vidal-Legaz, B., Martínez-Fernández, J., Picón, S. A., Pugnaire, F. I. 2013. Trade-offs between maintenance of ecosystem services and socio-economic development in rural mountainous communities in southern Spain: A dynamic simulation approach. Journal of Environmental Management, 13:280-297.

Vigerstol, K. L., Aukema, J. E. (2011). A comparison of tools for modeling freshwater ecosystem services. *Journal of Environmental Management, 92*:2403-2409.

Viviroli, D., Dürr, H. H., Messerli, B., Meybeck, M., & Weingartner, R. (2007). Mountains of the world, water towers for humanity: Typology, mapping, and global significance. *Water Resources Research*, *43*(7), W07447. doi:10.1029/2006WR005653

Zakkak, S., Kakalis, E., Radović, A., Halley, J. M., & Kati, V. (2014). The impact of forest encroachment after agricultural land abandonment on passerine bird communities: The case of Greece. *Journal for Nature Conservation*, *22*(2), 157–165. doi:10.1016/j.jnc.2013.11.001

Appendix 4.1 Input data and data sources for the various InVEST models considered in the present study.

Input data	Format (resolution)	Units	Data sources	Model
Land Use Land Cover (LULC)	Raster (30 x 30m)	-----	Rodrigues 2010	Carbon Storage & Sequestration; Sediment Retention; Water Yield; Nutrient Retention
Digital elevation model (DEM)	Raster (30 x 30m)	-----	ASTER GDEM (http://reverb.echo.nasa.gov/reverb/redirect/wist)	Sediment Retention; Nutrient Retention
Watersheds (WS)	Shapefile	-----	Atlas da Água (SNIRH)(http://snirh.apambiente.pt/)	Sediment Retention; Water Yield; Nutrient Retention
Sub-watersheds (sWS)	Shapefile	-----	Atlas da Água (SNIRH) (http://snirh.apambiente.pt/)	Sediment Retention; Water Yield; Nutrient Retention
Rainfall erosivity index (R)	Raster (250 x 250m)	$MJ*mm*(ha*h*yr)^{-1}$	Atlas da Água (SNIRH) (http://snirh.apambiente.pt/)	Sediment Retention
Soil erodibility (K)	Raster (500m)	$T*ha*h*(ha*MJ*mm)^{-1}$	Joint Research Center (http://eusoils.jrc.ec.europa.eu/library/themes/erosion/Erodibility/)	Sediment Retention
Average annual precipitation (PP)	Raster (1000 x 1000m)	Millimeters	Atlas Climático Digital da Peninsula Ibérica (http://opengis.uab.es/WMS/IBERIA/index.htm)	Water Yield;
Average annual reference evapotranspiration (ETo)	Raster (x m)	Millimeters	Global PET	Water Yield;
Plant Available Water Content (PAWC)	Raster (30 x 30m)	Fraction	Shape Solos Minho	Water Yield;
Soil depth (SD) as a proxy for root restricting layer depth	Raster (30 x 30m)	Millimeters	Shape Solos Minho	Water Yield;

Appendix 4.2. Values (grey column) extracted from the literature (white column) for the four fundamental carbon pools used to model carbon storage and sequestration in InVEST. Units are metric tons/hectare $Mg.ha^{-1}$. The land use categories of bare rock, urban and water were given a value of 0 for all carbon pools as recommended by InVEST user's guide.

Land Use	**C_above**		**C_below**		**C_soil**		**C_DOM**	
Agricultural fields	5	IPCC (2006)	6	IPCC (2006)	38	IPCC (2006)	5	IPCC (2006)
Oak forest	130	Garcia (2010)	15	Balboa-Murias et al. 2006	98	Rosario (2009)	6	Fernandes and Rigolot (2007)
Low scrubland	4	Bompastor et al. (2010)	10	Bompastor et al. (2010)	91	Bompastor et al. (2010)	1	Bompastor et al. (2010)
Tall scrubland	18	Bompastor et al. (2010)	11	Bompastor et al. (2010)	91	Bompastor et al. (2010)	3	Bompastor et al. (2010)
Pine	150	Garcia (2010)	66	Fernandes and Rigolot (2007)	116	Rosario (2009)	7	Rosa (2011)

Appendix 4.3 Input values and sources for Biophysical Table (water yield and sediment retention model).

LULC	Factor C[*1]	Factor P[*2]	Sedret eff[*3]	Root_depth[*4]	etk[*5]
agricultural fields	400	450	30	1500	650
oak forest	100	1	100	6000	1000
low scrubland	20	1	100	2000	398
tall scrubland	100	1	100	4700	398
Pine	50	1	100	4850	1000
bare rock	10	1	5	0	1
Urban	10	1	5	0	1
Water	0	1	5	0	1

[*1] Values for Factor C (cover and management factor) can be found in Pimenta 1999. Input for sediment Retention Model.

[*2] Values for Factor P (Support practice factor) were taken from Wischmeier and Smith 1978. Input for sediment Retention Model. We assumed that it did not varied between years.

[*3] Sediment retention efficiency values were based on InVEST suggested values. Input for sediment Retention Model.

[*4] Root depth is the maximum root depth for vegetated land use classes, given in millimeters Non-vegetated classes should be given a value of 0 (as suggested in InVEST guidelines). Values can be found in Canadell et al. 1996 and Silva and Rego 2004. Input for Water Yield Model.

[*5] Etk it is the plant evapotranspiration coefficient for each LULC. The values were multiplied by 1000 so that the final etk values are integers ranging between 1 and 1500. Etk values were based on InVEST suggested values. Input for Water Yield Model.

Appendix 4.4 Variation in local species richness from 1960 to 2007 (average number of species per 25 ha cell; n cells = 450).

Species group	Mean variation in species number
Forest plants	-0.49
Forest birds	-0.21
Shrubland plants	0.43
Shrubland birds	0.06
Agricultural plants	-3.95
Agricultural birds	-0.19
Ubiquitous plants	-0.11
Ubiquitous birds	-0.04

Appendix 4.5 Calculated carbon storage for each land cover for aboveground and belowground biomass, soil and dead organic matter for 1960.

LULC	LULC name	N pixels	Area (ha)	value C / pixel	Mg/ha	Total landscape(Mg)
1	Urban	650	58.5	0	0	0
2	Agri	13476	1212.8	4.86	54	65493
3	Oak	11355	1021.9	21.6899	241	246289
4	Rock	33390	3005.1	0	0	0
5	Lowshr	37006	3330.5	9.54	106	353037
6	Tallshr	9652	868.6	11.07	123	106848
7	Pine	2548	229.3	30.51	339	77739
8	Water	227	20.4	0	0	0
	TOTAL		9747		863	849407

Appendix 4.6 Calculated carbon storage for each land cover for aboveground and belowground biomass, soil and dead organic matter for 1990.

LULC	LULC name	N. Pixels	Area (ha)	value C / pixel	Mg/ha	Total landscape (Mg)
1	Urban	1046	94.1	0	0	0
2	Agri	6841	615.6	4.86	54	33247
3	Oak	12126	1091.3	21.69	241	263012
4	Rock	35305	3177.4	0	0	0
5	Lowshr	38750	3487.5	9.54	106	369675
6	Tallshr	12387	1114.8	11.07	123	137124
7	Pine	1620	145.8	30.51	339	49426
8	Water	230	20.7	0	0	0
	TOTAL		9747		863	852484

Appendix 4.7 Calculated carbon storage for each land cover for aboveground and belowground biomass, soil and dead organic matter for 1990.

LULC	LULC name	N. Pixels	Area(ha)	value C / pixel	Mg/ha	Total landscape
1	Urban	1103	99.2	0	0	0
2	Agri	6228	560.5	4.86	54	30268
3	Oak	11775	1059.7	21.6899	241	255399
4	Rock	34868	3138.1	0	0	0
5	Lowshr	35703	3213.2	9.54	106	340607
6	Tallshr	16820	1513.8	11.07	123	186197
7	Pine	1580	142.2	30.51	339	48206
8	Water	229	20.6	0	0	0
	TOTAL		9747		863	860676

Chapter 5. Ecosystem services: the opportunities of rewilding Europe

Abstract:

Halting the degradation and restoring the full capacity of ecosystems to deliver ecosystem services is currently a major political commitment in Europe. Although still a debated topic, Europe´s on-going farmland abandonment is seen as an opportunity to launch a new conservation and economic vision, through the restoration of natural processes via rewilding as a land management option. Despite the ecological interest of restoring a wilder Europe, there is a need to generate evidence-based arguments and explore the broad-range impacts of rewilding. In this chapter we contribute to the on-going debate on rewilding by first analyzing the spatial patterns of ecosystem services in both the EU25 and in wilderness areas. We subsequently quantitatively explore the supply of ecosystem services in the top 5% wilderness areas, on agricultural land, and on land projected to be abandoned, at the extent of the Iberian Peninsula. We determine that high quality wilderness is often associated to high supply of ecosystem services, mainly regulating and cultural, specifically in mountain regions. Assuming that high quality wilderness is a good proxy for rewilding, our results suggest that rewilding efforts throughout Europe will enhance the capacity of ecosystems to supply high regulating and cultural ecosystem services, such as carbon sequestration and recreation.

Keywords: benefits, biodiversity, ecosystem services, human well-being, rewilding, wilderness

5.1. Introduction

Ecosystem services have been defined as the benefits humans derive from nature through a set of ecosystem functions. The Millennium Ecosystem Assessment (MA) was the stepping stone in providing a conceptual framework for ecosystem services as well as, assessing the consequences of ecosystem change for human well-being and the overall global trends. Since its publication (MA 2005), multiple classification schemes for ecosystem services have been proposed, such as The Economics of Ecosystem Biodiversity (TEEB 2012) and, more recently, the CICES (Common International Classification of Ecosystem Services). Adopted by the European Commission for the new Biodiversity Strategy for 2011-2020 (Maes et al. 2013), the CICES categorizes ecosystem services into 3 groups (Haines-Young and Potschin 2013): provisioning (e.g. food, fiber, fuel and water), regulating and maintenance (e.g. air quality, water and soil regulation, natural hazard regulation, climate regulation and disease control) and cultural (e.g. recreation and spiritual).

Although society can easily perceive provisioning ecosystem services such as crops, fish and freshwater, which are all direct benefits to humans, others, such as pollination, erosion control and climate regulation are less tangible. However, directly or indirectly, all ecosystem services underpin environmental and human well-being, economy, and businesses (MA 2005). Many services are not traded in the conventional markets and hence, their economic values remain invisible, tending to be undervalued and consequently overexploited (de Groot et al. 2012). Yet, once lost, replacement can be costly. Wetlands, for example, provide numerous regulating services (e.g. water purification and flood/storm protection), which are unnoticed, in contrast to provisioning services (e.g. timber and food), but highly valuable since degradation can lead to high replacement costs (Reed et al. 2013). For instance, in New York City, the replacement cost for water purification services was estimated at $6 to $8 billion in capital costs for the installation of a water treatment plan whereas, the investment in conservation measures to protect the wetlands within the watershed, that will purify the water, was estimated at US $1.5 billion (Kenny 2006).

Throughout the world, ecosystem services have been used as a tool in conservation and development as well as poverty alleviation (Tallis et al. 2008). However, many conservation efforts have been unsuccessful due to human mismanagement of ecosystem services. The awareness that ecosystem services affect human well-being and economic development has resulted in their integration in policies and government strategies and the most recent EU Biodiversity Strategy. This new strategy sets goals to halt both biodiversity loss and the degradation of ecosystem services. In particular, it includes the protection of wilderness, specifically old growth forest. Today, 45% of Europe´s land cover is forest (1 billion ha) but only 4% is undisturbed forest (6 million ha). Protecting these ecosystems is important as they support high quality ecosystem services, such as recreation and air quality (Maes et al. 2012a). Increasing these percentages could greatly improve the supply of ecosystem services provided by natural habitats of no human influence. This new conservation strategy goes hand in hand with a fairly recent initiative, “Rewilding Europe” which aims to rewild one million hectares of land by 2020. In particular, the emergence of the rewilding topic is seen as an opportunity for rural areas which have undergone land abandonment throughout the past decades. However, we have yet to determine if rewilded areas will promote the supply of ecosystem services fundamental for human well-being.

In this chapter, we first investigate the supply and spatial distribution of ecosystem services on a pan-European scale before comparing it with the occurrence of wilderness areas. We then use the supply of ecosystem services in wilderness areas as a proxy for areas that are projected to undergo land abandonment and rewilding. Finally, we discuss the various economic and ecological benefits of rewilding in Europe.

5.2. Europe and Ecosystem services

5.2.1. Current supply of ecosystem services

Ecosystems provide a number of essential services underpinning all human life and activities. However, the continuance of the various ecosystem services is only possible through recognition of ecosystems multiple functions integrated in management strategies. To manage for multiple ecosystem services we need to map and identify the spatial synergies and trade-offs between services (Raudsepp-Hearne et al. 2010; Maes et al. 2012a). In doing so, we are able to identify ecosystems supporting high level of services and biodiversity (Chan et al. 2006). Along the years, the number of studies mapping ecosystem services and scenario building has grown, informing both planners and decision makers, prioritizing the protections and management of ecosystems (Chan et al. 2006) and additionally, delineating cost-effective measures (Egoh et al. 2008; Naidoo et al. 2006).

The integration of ecosystem services into current conservation strategies ensures future sustainability. However, the integration of ecosystem services in Europe´s conservation strategies has only recently scratched the surface. In the EU Biodiversity Strategy for 2020, the need for spatial assessment of ecosystem services has been included as one of the key actions. Under Action 5 all EU Member States are required to map and assess the state of ecosystems and their services by 2014, addressed by the Working Group and Mapping and Assessment of Ecosystems and their Services (MAES). The results of this action will help inform policymakers but also contribute to the assessment of the economic value of ecosystem services which are to be integrated into the accounting and reporting systems at both EU and national level by 2020 (European Commission 2011a).

The spatial mapping of ecosystem services throughout Europe forms the framework for the first part of this study. A total of 6 ecosystem services, represented by 9 indicators,

Table 5.1 List of the Ecosystem Services and corresponding indicators used in the study (adapted from Maes et al. 2011).

Category	Service	Indicator	Unit	Description / Benefit
Provisionning	Food provision	HANPP	gC/m2/yr	Human Appropriation of Net Primary Production (cropland and grassland in this study)
	timber provision	total stock of timber	m^3/há	Production for fuel, construction and paper. Forest connectivity.
	freshwater provision	Surface Water Flow (QFS)	Mm	Renewable freshwater provision.
Regulating	Climate regulation	Carbon stock	ton/há	Above- and below-ground carbon stored in living plant material.
		Net Ecosystem Productivity (NEP)	mg/m^2/year	Carbon sequestration.
	Water regulation	Nitrogen retention	%	Capacity of ecosystems to retain and process excess nitrogen
		Soil infiltration capacity	Mm	Annual summed infiltration capacity of water
	Air quality	Deposition velocity of Nox	cm/s	Capacity of ecosystems to capture and remove air pollutants
Cultural	Recreation	Recreation potential index (RPI)	N/A	Capacity of ecosystems to provide recreational services

were considered for this analysis (Table 5.1). In order for each ecosystem service to contribute equally to the analysis, and following the method of Petter et al. (2012), we standardized the data by reclassifying each service into a quantile split, producing a range of scores from 1 to 5 (five meaning high supply of a specific service). We then summed the 9 indicators to produce a map of "total" ecosystem services supply across Europe (Fig 5.1a). We used HANPP (Human Appropriation of Net Primary Production) data presented in Haberl et al. (2007) study, as one of the indicators for provisioning services. For HANPP, we extracted only the values within arable and cropland.

Our findings show a non-surprising spatial correlation between the supply of services and the European land-cover (Fig 5.1b). Low stocks for ecosystem service supply appear mainly around urbanized and densely populated areas and in arable land, e.g in central and eastern Spain, Southern Romania, Eastern UK, and Denmark. However, low total supply of services doesn't mean a low quality of the supply of individual services. For example, even if food production were at their highest level in some areas, if that is the only services provided, such area would appear in the low range of the map. High ecosystem supply includes mainly pastures, forests and (semi) natural areas, such as the North-west Iberia, Scandinavia, central France, and central Romania.

Overall, we also observe that key areas of ecosystem service supply in Europe coincide with mountain regions (Fig. 5.1a), mainly consisting of forest and (semi)-natural areas (Fig. 5.1b). As a matter of fact, dense forest cover in mountain areas, and regions rich in wetlands (including mountain and lowlands) have been previously assessed as regions of high ecosystem service supply (Maes et al. 2012b). Nonetheless, this does not mean that non-mountain areas do not supply valuable ecosystem services. Intensively managed agro-ecosystems are generally associated to flatter more fertile regions in Europe. Approximately, 45% of Europe´s land area is under agriculture (EEA 2006) and, considered essential food source providers.

Changes in human demand for services associated with specific land uses have shown diverging trends in Europe, varying between regions. In general the demand for crops, timber (mainly in northern countries), freshwater, and recreation has increased in the last 50 years while livestock production and wild foods supply have followed a decreasing trend throughout much of Europe´s rural areas (Harrison et al. 2010). During this period the quality of some ecosystem services have improved, mainly those services associated to forest ecosystems in mountain systems (i.e. timber production, freshwater provision, erosion and natural hazard regulation, and recreation), partly due to a decrease in human pressure in remote areas of the continent (Harrison et al. 2010).

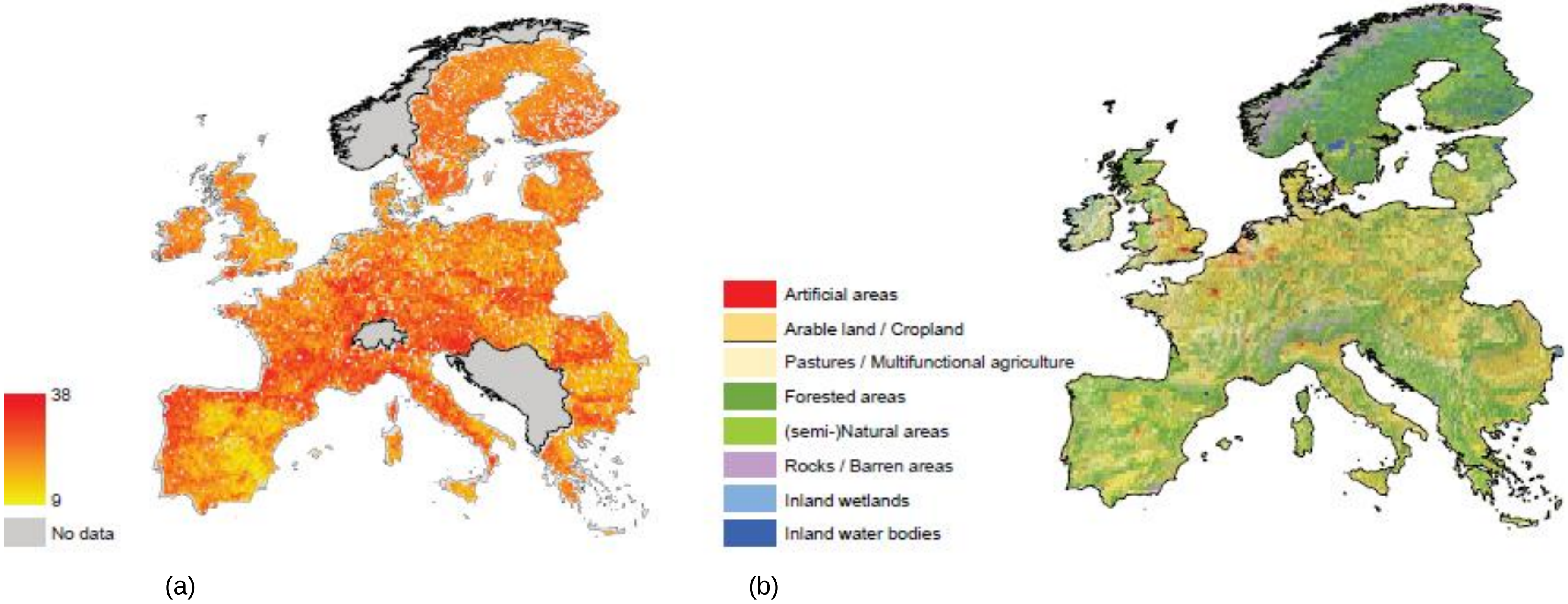

Figure 5.1. Ecosystem services and land-covers in Europe. A) Sum of the quantile splits of all indicators used in the analysis. With each quantile split, services can reach values between 1 and 5, 5 being the highest. By summing all 9 indicators (see Table 5.1), the gradient potentially varies between 9 and 45 but, de facto, the maximum and minimum values are 9 and 38. (Method detailed in the text – Maes et al. 2011); B) Map of European land-covers based on the Corine Land Cover data base of 2006 (EEA 2013).

5.3. Threats and opportunities

Society has along the years taken for granted the services that are provided by natural ecosystems, exerting pressures due to human demand. This has lead to the unsustainable use and degradation of their natural functions and processes, disregarding the fact that ecosystems are fundamental in providing goods and services essential to human well-being and economy (TEEB 2010). The Millennium Ecosystem Assessment was the first scientific appraisal of the global condition and trends of ecosystems and their services. The assessment revealed that in the last 50 years, increased anthropogenic pressures have changed ecosystems globally to meet rapidly growing demands for provisioning services, such as, food, fresh water, timber, fiber, and fuel (MA 2005). This led to the decline in other ecosystem services, mainly regulating services (e.g. flood regulation and nutrient cycling) (MA 2005). According to the MA (2005), approximately 60% of ecosystem services examined are being degraded or used unsustainably and expected to increase in the context of climate change. In Europe, the state of most ecosystem services have been categorized as degraded or mixed. However, timber production associated to forests and mountains, freshwater provision, natural hazard regulation and recreation have all improved (Harrison et al. 2010).

Land conversion has been identified as the primary driver of ecosystem change (Klein-Goldewijk 2011). Other drivers of substantial influence include climate change, nutrient application to agricultural systems, biological invasions and diseases, as well as indirect drivers such as socio-economic, political and demographic changes (MA 2005). Years of unsustainable farming practices and mismanagement through agricultural intensification have resulted in a highly fragmented Europe, impacting biodiversity and the supply of ecosystem services (Schrötter et al. 2005; Egoh et al. 2008). In particular, the unsustainable management of agro-ecosystems has contributed to the loss of habitat and biodiversity, soil erosion and nutrient runoff (Dunbar et al. 2013).

In an attempt to halt further degradation or loss of ecosystem services and biodiversity, Europe has adopted the new Biodiversity Strategy Plan for 2020. The new 2020 strategy has a focus on ecosystem services, highlighting their natural and economic value and the importance of maintaining and restoring them (European Commission 2011a). The EU Strategy delineates a total of six Aichi Targets which aim at reducing pressures on biodiversity and ecosystem services, and mapping and assessing the state of ecosystems and their services to incorporate their full value into national and EU accounting and reporting systems. For instance, target 2 (Action 6) promotes the restoration and the use of green infrastructures (i.e interconnected network of ecosystems, such as wetlands and woodlands) through incentives, restoring 15% of degraded ecosystems through both EU funding and Public Private Partnerships (European Commission 2011a).

Acknowledging that both ecosystems and the services they provide are at risk due to human pressures, and that humans rely directly and indirectly on them for their well being sets the frame for both a paradox and an opportunity, depending on which service is referred to. On the one hand, some services put ecosystems at a risk of overexploitation and unsustainable use, while, on the other hand, the supply of other services can become an incentive to preserve or restore ecosystems. Through the restoration of green infrastructures, natural areas will be reconnected, thus improving ecosystem functionality. The introduction of financial incentives is also seen as a cost-effective measure of investing in nature through the protection and restoration of services, reducing costs related to treatments and potential natural hazards (de Groot et al. 2013). For example, the Danube river wetland restoration project, has promoted the concept of payment for ecosystem services (PES) to restore ecosystem capacity to retain flood waters, decreasing flood impacts, ultimately reducing flood costs while improving benefits to nature, people and local economies (ICPDR 2010). Besides its growing incorporation in government policies, the use of PES (payment for ecosystem services) has also been applied at the business level. For example, the Nestlé Waters Programme, has created a union with local farmers in northeastern France to adopt farming practices that reduce nitrate pollution, providing high quality drinking water while compensating farmers (Bishop and Timberlake 2007).

Though this type of economic incentives make sense in inhabited areas, where farmers, for example, could be asked to apply agri-environmental schemes to promote biodiversity and ecosystem services, a different approach can be envisioned in areas where the land is progressively relieved from agriculture and abandoned (Merckx and Pereira submitted). Over the last couple of decades, some European landscapes have witnessed the transition from extensive agriculture and semi-natural grasslands to abandoned land. This phenomenon is driven by several factors, which include the lack of economic opportunities, shifts in values and attitudes among people, a decreased dependency of local provisioning services (e.g. food and fuel) and environmental constrains, amongst others (MA 2005; MacDonald et al. 2000; Pereira et al. 2005). Along the years these drivers have crossed paths, joining forces, exacerbating rural exodus, leading to land abandonment, for the most part in marginal regions.

Although not a recent phenomenon, this rapid land use change has caught the attention of conservationists, economists, scientists, and policy makers. However, there still remains a large disagreement between those who see abandonment as a threat and those who see this trend as a window of opportunity for those ecosystem services which in the past were sacrificed due to human demand. These contradictory viewpoints create challenges in delineating land management plans as well as adequate policies. In the past, policy measures, such as the Less Favored Areas (LFA) have been implemented in an attempt to attenuate rural migration and promote agricultural activity in less productive and remote regions of Europe (Dax 2005). For instance, approximately 92% of the total mountain areas in the EU27 have been designated as LFA (EEA 2010). However, efforts have gone unnoticed, which is clearly represented by the continuous decreasing trend of agricultural area and rural population, both projected to continue in the future. Presently, only 14% of the total farmers in the EU 25 benefit from compensatory payments under the LFA (European Commission 2008), while, the new proposal for the Common Agricultural Policy (CAP) has included the restoration and conservation of ecosystem services under the rural development pillar (European Commission 2011b). Considering that past attempts have been fruitless in halting abandonment in remote and less productive rural systems, and funded on the socio-economic challenges of maintaining extensive agriculture active, we can now look at the opportunity of rewilding these areas throughout Europe as a mean of generating environmental, social and economic benefits from ecosystem services provided by restored and self-sustainable ecosystems. By using present wilderness areas as a proxy we can determine the potential supply of ecosystem services through the promotion of rewilding in those areas projected to undergo abandonment.

5.4. Wilderness and Ecosystem services

5.4.1.Wilderness

Wilderness areas have been defined as large natural areas, unmodified or slightly modified governed by natural processes, with no human intervention, infrastructure or permanent habitation present (Wild Europe 2012). In Europe, core wilderness areas are mainly concentrated in high altitude areas, predominantly mountain areas. Nordic mountains represent the highest proportion (28%) of wildest areas, followed by the Pyrenees (12%) the eastern Mediterranean islands and Alps (9 %), and British Isles (8 %) (Carver 2010). However, remnants can also be found throughout much of the continent, where anthropogenic interference has slightly altered the natural ecological conditions (Carver 2010). Note that, though the concept of wilderness is now commonly understood by the scientific community in Europe, the definition of wilderness will depend on the metrics chosen and, as a result, its spatial distribution can vary from one study to another. Currently, there are several maps on potential wilderness in Europe, however, in the present study we chose to use Carver's (2010) quality wilderness index.

Many wild areas are under threat and represent a small proportion of the European continent. For example, although forests make up 33% of Europe´s land cover, only 5% (9 million hectares) is considered wild. Currently, numerous organizations are focusing on the expansion of wild areas through the restoration of adjacent areas, while protecting remaining pristine regions. PAN Parks is one example of how the creation of a network has resulted in the protection of wilderness throughout Europe.

Wild ecosystems are healthy systems that provide a wide range of ecosystem services. They are stable and self-sustainable, able to maintain their structure, function and resilience over time (Costanza and Mageau 1999). They play an important role in protecting services such as, air quality, freshwater provision, and supporting wildlife, including charismatic species, such as bisons and bears, that are reliant on wilderness areas (Russo 2006). Wild ecosystems also have the capacity to supply higher quality services than other types of systems. For example, there is higher carbon storage capacity in undisturbed forest, peatland and wetland (Schils et al. 2008), subsequently providing additional environmental benefits (e.g. biodiversity, water storage and water quality).

Moreover, wilderness areas provide a range of social and economic benefits. Several programs have integrated the use of wild areas to address urban issues such as youth at risk, youth development and rehabilitation (Hill 2007), and recognized as a cost-effective form of healthcare. In addition, wilderness inspires educational programs. Wilderness areas also provide spiritual benefits, such as, solitude, places of inspiration, a calm environment, and recreation/tourism (Heintzman 2013; Ewert et al. 2011). These social benefits can give birth to employment opportunities and thus generate income. For example, the Oulanka National Park in Finland brings €14 million per year to the local economy and employs 183 individuals (Huhtala et al. 2010).

5.4.2. Mapping and Methods

We used Carver's (2010) wilderness quality map and the same quantile approach described earlier in section 3.2 to produce a gradient of wilderness quality with qualitative values ranging between 1 and 4 (4 meaning the highest wilderness quantile and high supply of ecosystem services). We then proceed to grouping the ecosystem services into provisioning, regulating, and cultural services and followed the same splitting approach for each category of service. The ecosystem services maps were then overlaid with the wilderness map. To determine the relationship between gradients of both ecosystem services supply and wilderness quality, we display the overlay of high and low wilderness with high and low supply of ecosystem services (Fig. 5.2 a, b, c and d). Furthermore, we used the projections of the CLUE model (Verburg and Overmars 2009) to assess the potential change in the provision of ecosystem services with scenarios of land abandonment and rewilding in Europe for 2030. We considered as potential land abandonment and rewilding the cells classified as arable land, pasture, irrigated arable land, permanent crops in 2000 and classified as (semi)-natural vegetation, forest, recently abandoned arable land and recently abandoned pasture land in 2030 common to all four EURURALIS scenarios. For quantitative comparisons, we calculated the mean provision of ecosystem service (per km^2) in agricultural areas (based on the 2000 land use map, in Verburg and Overmars 2009), in the top 5% high quality wilderness, and in the areas currently under agricultural use but projected to become abandoned by 2030, in the Iberian Peninsula (Table 5.2). Significant differences between the distributions of the values for each type of land-use were tested using a Kruskal Wallis test. Finally, the ratio between the average supply of each indicator, and the common highest value for these indicators, were calculated in all three land-uses type studied in order to compare the relative supply of ecosystem services in each case. All mapping and data extraction were done using ArcGIS version 10.3, while the statistical analysis was done using R version 2.15.3.

5.4.3. Analysis

When looking into the total ecosystem service supply in Europe (Fig. 5.2.a), we observe that it is lower (see Color Key 1 & 3) in densely populated areas and in agricultural areas, while it is higher (see CK 2 & 4) in forested and mountain areas. In addition, we determined that some of the high wilderness areas (see CK 3 & 4) are associated to those regions, mainly mountain systems, supplying high ecosystem services (see CK 4). The same qualitative analysis was done separately for provisioning, regulating, and cultural services (represented by an indicator of recreational services).

In comparison with other categories of ecosystem services, the overlay of provisioning services and wilderness (Fig. 5.2.b) presents relatively large areas of high supply of services and low wilderness (see CK 2, e.g. in France, Benelux and Germany), along with areas of low service supply and high wilderness (see CK 3, e.g. Northern Scandinavia). This is not surprising since wilderness areas are typically associated with low to no extraction of natural resources. However, Southern Scandinavia provision a number of important services, such as cereal production (Kettunen et al. 2012) which are also more densely populated than in the north. There are nonetheless some representations of high provisioning services in areas of high wilderness quality (see CK 4), mainly associated to mountain regions, appearing in mountain regions (e.g. some areas of the Alps and Apennines).This can be due to the occurrence of large quantities of resources for some provisioning services (i.e. timber and freshwater) in mountain regions, which also happen to be wilder than the rest of Europe.

The spatial distribution of regulating services coincided relatively well with wilderness (Fig.5.2.c), with large areas of Europe containing both high supply of services and high degrees of wilderness (see CK 4, e.g. Northern Iberia, Austria). Most of the continent is still represented by areas of both low regulating services and low wilderness (see CK 1, e.g. Eastern UK, Poland), which also coincides with agricultural areas (Fig 5.2.b).Interestingly, several areas of high service supply and low wilderness appear on the map (see CK 2, e.g. Western France and Ireland). Finally, for recreational services (Fig. 5.2.d), we found a predominance of either areas of low service and low wilderness (see CK 1), or areas of high wilderness and high service (see CK 4). The pattern for areas of low wilderness and high service supply (see CK 2) is completely different than for the other categories of services, with a rather small representation on the map. We also observe a consistent amount of areas with high wilderness but low recreation potential (see CK 3). This particular result may be due to the challenges associated to measuring the capacity and flow of benefits related to cultural services. For example, one may have an ecosystem of extreme beauty or wilderness quality, however, if they are not accessible, the flow of recreation and other cultural services is low. On the other hand, one may have a less natural area but easily accessible due to distance to human infrastructures such as roads. These, somehow contradictory, metrics can explain the observed pattern in the case of cultural services.

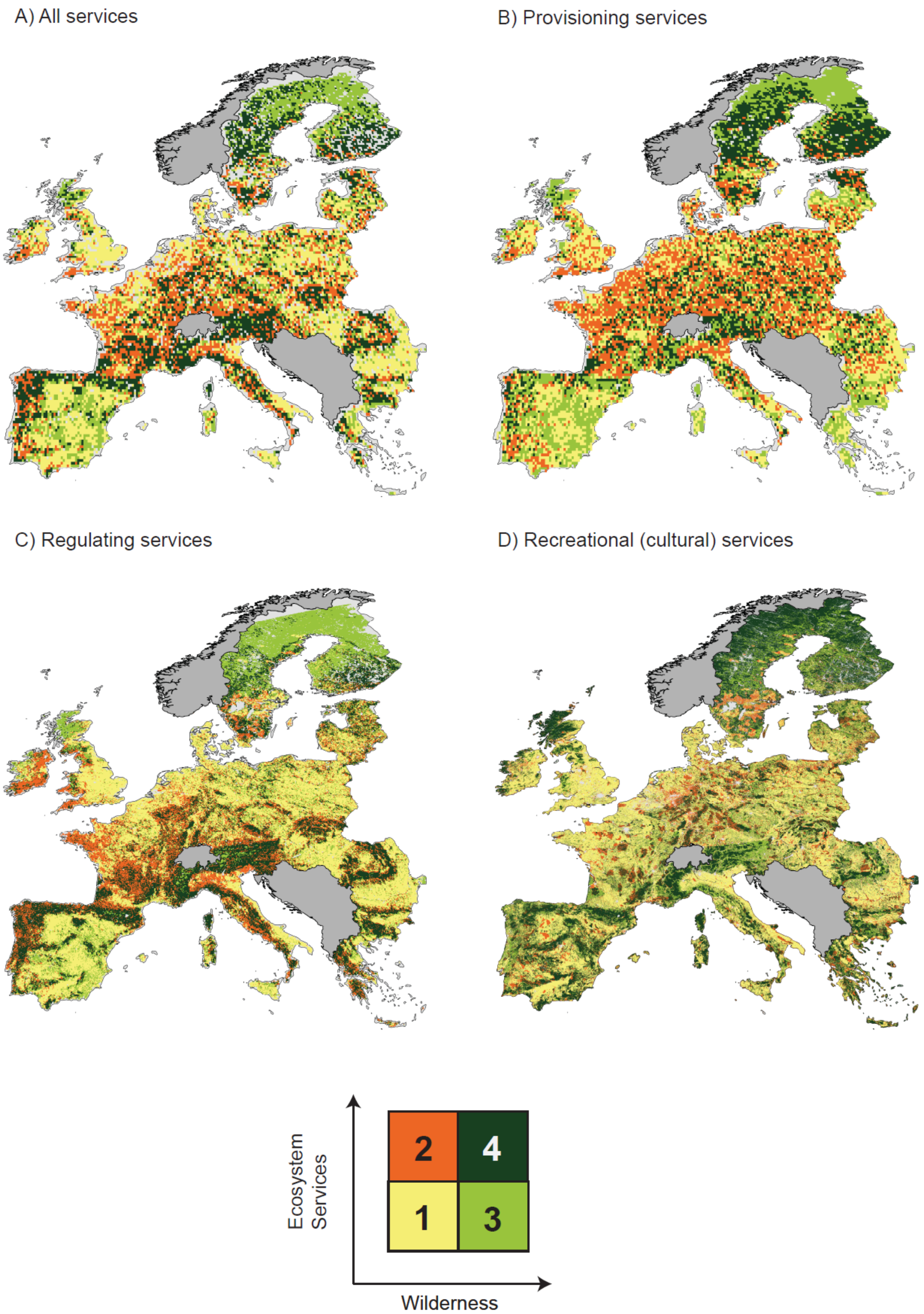

Figure 5.2. Ecosystem services and wilderness in Europe. For each map, the quantile splits of ecosystem services and wilderness were overlaid to present a gradient of both wilderness and service supply. For an easier representation, the values were grouped into "low" (bottom 50%) and "high" (top 50%) for both metrics and then grouped, e.g. low supply of services and low wilderness (see color key on the figure). A) All indicators for all services versus wilderness; B) Indicators of provisioning services versus wilderness; C) Indicators of regulating services versus wilderness; and D) Recreational service versus wilderness. (See Table 5.1 for a details of the indicators used). Sources: (Maes et al. 2011; Carver 2010).

Taken as a whole, regulating and cultural service are often associated to high wilderness areas (Fig. 5.2. b and c), particularly mountain systems. Mountain ecosystems cover approximately 41% of Europe´s territory, providing various services due to their multifunctionality. They are generally referred to as “water towers” important for lower elevation ecosystems, irrigation, industry, hydropower and supply freshwater to more than half of the human population (Viviroli et al. 2007). In mountain systems we can find a high proportion of habitat types with favorable conservation status (EEA 2010), playing a key role in provisioning many ecosystem services and maintaining ecological processes (Harrison et al. 2010). These mountain habitat types include natural grasslands and mountain peatlands contributing to flood prevention, soil erosion, climate stability, and recreational services, such as bird watching (Silva et al. 2008). Specifically, peatlands store large quantities of carbon and have played a fundamental role in climate regulation and are critical for water regulation. Meanwhile, grasslands are habitat to a large number of species, such as wild pollinators (Kremen et al. 2002), which makes them essential in underpinning biodiversity and ecosystem services. These ecosystems' adaptive capacity to both temperature and altitude gradient have made them ecological hotspots of biodiversity and endemic species. In Europe, the highest number of endemic species can be found in the Alps and the Pyrenees (Väre et al. 2003).

Mountain systems also play a key role in regulating erosion and natural hazards influenced by vegetation cover (Körner 2002), more specifically forested land. Forests make up 41% of mountain systems (Körner et al. 2005) and can be regulators of natural disasters as the soils of mature forests have high infiltration rate, thus reducing peak flows and floods (Maes et al. 2009). They also provide a range of services such as carbon sequestration, air quality regulation, timber for fuelwood and non-timber products (game and medicinal plants), and climate regulation (Harrison et al. 2010; Maes et al. 2012a). Moreover, forests and mountain systems supply cultural services, holding spiritual and religious value to local inhabitants, and are main recreation and ecotourism attraction (Price et al. 1997). Though mountain systems are tagged as major suppliers of ecosystem services, they also are fragile systems which recover slowly or not at all from disturbances which consequently decreases ecosystem function, impacting lowland areas and overall, human well-being (Körner et al. 2005).

5.5. Rewilding and ecosystem services

5.5.1. Ecosystem services and scenarios of rewilding

Although, many reports have outlined the positive benefits of wilderness and wild areas, the potential gains and losses of ecosystem services through the promotion of rewilding is still understudied. Our analysis contributes to reducing this lack of data by quantifying the biophysical potential of a system to produce benefits after rewilding has occurred, by comparing it with the current supply in other agricultural areas in the Iberian Peninsula and using wilderness as a proxy for the future supply of services. Generally, we speculate that by increasing the size and connectivity of high quality wilderness, we will have an increase in the supply of ecosystem services associated with those habitats (that is, if the abandoned land is restored to a self-sustainable state, either naturally or via assisted restoration). Concomitantly, the supply of services associated with agricultural land-uses might decrease.

By comparing the distribution of the values, at the extent of the Iberian Peninsula, for the supply of indicators of ecosystem services, between “areas currently cultivated”, “areas cultivated but predicted to be abandoned”, and “areas within the top 5% of Iberian wilderness”, we were able to predict a potential increase or decrease in the supply of the studied services.

We determined significant differences in the supply of ecosystem services and the different land use intensities (Table 5.2). We found HANNP values to be significantly higher in present agriculture areas (279.03) than in both land projected to be abandoned (271.15) and top 5% wilderness areas (224.12). HANNP has been identified as an indicator measuring changes in biomass flows in ecosystems and the provision of important ecosystem services, as a result of land use change (Erb et al. 2009). Our results are not unexpected as human consumption of food is greater in agricultural areas than all other land uses. For deposition velocity of NOx, an indicator of air quality, values were higher in wilderness (0.42 cm/s) in comparison to the other land use intensities, land projected to become abandoned (0.28 cm/s) and present agriculture (0.07 cm/s). We can consider that air quality improves in wilderness areas attributed to the capacity of the ecosystems to capture and remove air pollutants. Increasing wilderness areas thus leads to increases in air quality.

Nitrogen retention was highest in wilderness (3.04%) followed by agriculture (2.69%) and abandoned land (2.46%). While, soil infiltration capacity was greatest in recently abandoned land (32.15 mm), followed by wilderness (15.99 mm). Both nitrogen retention and soil infiltration are indicators of water regulation. We can assume that water regulation, improves with increased vegetation dynamics, as high values for these indicators were mainly represented in both top wilderness and projected abandoned land. Both timber and recreation values where highest for wilderness (210E+05, 0.43) and abandonment (1.34E+05, 0.25). Nonetheless, these results may be misleading, specifically for timber, since values calculated for wilderness only include the top 5% of wild regions which is expected to be highly protected. Another explanation for timbers high values in the top 5% wilderness may be due to the low resolution of the data.

For carbon stock and net ecosystem productivity we found a higher supply in land projected to be abandoned (74.81 ton/ha and 7.36E+5 mg/m2/yr) than those areas designated as wilderness (63.19 ton/ha and 6.99E+5 mg/m2/yr) (Table 5.2). Thus, the abandonment of those areas would support wilderness areas by increasing productive areas for both carbon stock and net ecosystem productivity, playing a fundamental role in climate regulation. Similar findings were found for freshwater quality, with higher values in land projected to become abandoned (244.77mm) than on wilderness areas (156.08mm), values fairly close to present agricultural areas (151.05mm). These findings support the premise that releasing land from agricultural activity, is an opportunity to increase or maintain high stocks of ecosystem services, in particular water quality, carbon storage and net ecosystem productivity (Fig.5.3; Table 5.2).

Although these results exemplify positive trends for the majority of the indicators, these results are to be read with caution as the transition from "recently abandoned" to "rewilded" is not fast, simple, or even guaranteed. Thus the supply of ecosystem services during the early stages after land abandonment can be hard to predict and complex. A recent study revealed that decreases in land use intensity, primarily the abandonment of mountain grasslands, lead to initial decreases in net ecosystem exchange of CO_2 (Schmitt et al. 2010). The impacts of nitrogen storage following abandonment are also not well understood. We do know that nitrogen storage in younger grasslands is known to be lower than older grasslands (Deng et al. 2013). Nonetheless, we can speculate that the benefits of releasing land from agriculture outweigh those of present agriculture activity. Recent studies have confirmed that the complexity of an ecosystem, which includes but not limited to its vegetation dynamics, the age and the distance and the extent of fragmentation are all elements which influence the supply of ecosystem services and biodiversity (Vanacker et al. 2014; Ferraz et al. 2014; Grêt-Regamey et al. 2014). And by restoring habitats we are increasing the landscapes multifunctionality through its services.

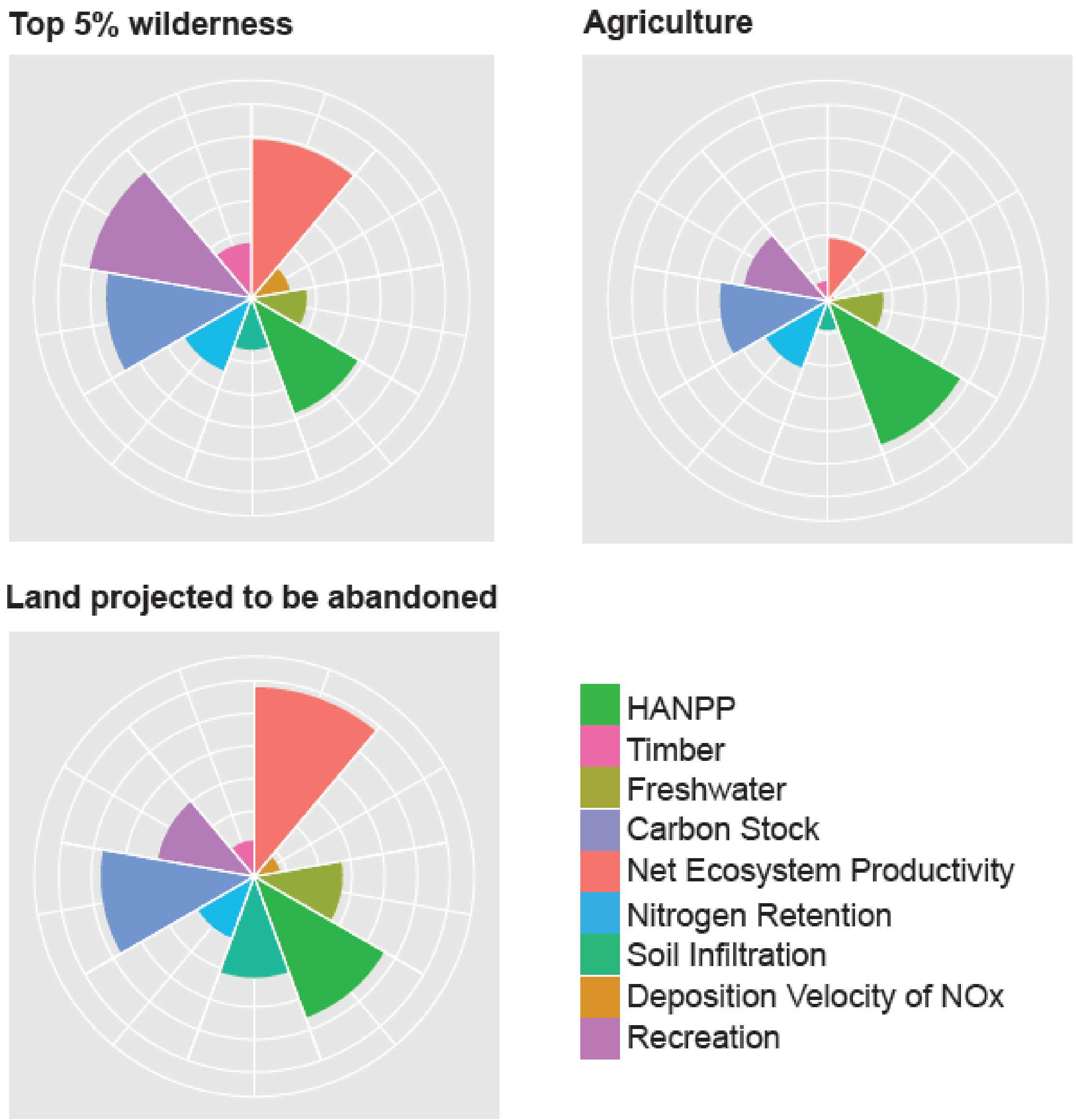

Figure 5.3 Quantitative assessment of the ecosystem services provided by the top 5% wilderness areas, present agricultural area and land projected to be abandoned in 2030 in Europe.

This analysis is based on projections of supply of service on areas where land abandonment or rewilding could potentially occur within the decades to come and so far is geographically limited to the Iberian Peninsula. Yet, rewilding has already occurred in some areas of Europe, either naturally or with assisted restoration and it is worth looking into the supply of ecosystem services in those "pilot" areas, though information on their benefits are still scarce. The earliest documented case of rewilding pioneered in the Oostvaardersplassen wetlands in the Netherlands, in the 70s. The introduction of wild herbivores, such as the red deer, Heck cattle and Konik ponies naturally substituted the ecological function of the now extinct aurochs and tarpans (Birdlife International 2011). Today, it is one of Europe´s leading wetland regions, where ecological restoration increased natural areas creating a win-win situation promoting regulating, cultural services and biodiversity while generating economic benefits through the promotion of tourism. However, there are still some limitations to this project as the perseverance of its success has been attributed to on-going human intervention along the years. In this section we present an additional number of case studies in which the promotion of wild areas and/or rewilding has generated environmental and economic benefits for local communities, landholders and society in general.

5.5.2. Regulating Benefits

In Lowland England, studies on different land use management options have shown that the cost and benefits of changes in ecosystem services from rewilding outweigh those from arable and dairy farming (Natural England Commissioned Report 2012). In the Upland UK estimates show that managing the land for carbon storage and sequestration through the restoration of peatlands may be more profitable than pastoral activities (Reed et al. 2013). As a matter of fact, peatlands, in Scotland, have been valued between €49 million to €196 per annum for carbon sequestration (McMorran et al. 2006).

The promotion of rewilding with forest regeneration on abandoned farmland can also be a major contribution to the mitigation of climate change through the sequestration and storage of carbon. It has been estimated that within the Natura 2000 network, commercial and wild forest habitats generate the highest carbon value estimated at €318.3 and €610.1 billion, in 2010 followed by grassland systems ranging between € 105.6 and €196.5 billion (ten Brink et al. 2011). In the Carpathians, the protection of old growth forest will generate a funding of €26 million through carbon offsets, providing regional economic relief (Ten Brink et al. 2011). In the Hoge Veluwe Forest, a protected area of the Netherlands, total economic benefit generated by forests is €2000 ha/year, for the following services; wood production, supply of game, groundwater recharge, carbon sequestration, air filtration, recreation and nature conservation. This value is calculated to be three times higher than adjacent agricultural land (Hein 2011).

Although there is still a lack of available information on the economic value of water purification at the EU level, studies confirm that cities such as Berlin, Vienna, Oslo and Munich benefit from the natural treatment from ecosystems in protected and non protected areas, with annual economic benefits ranging between €7 and €16 million for water purification and €12 and €91 million for water provision per city (ten Brink et al. 2011. In the archipelago of the Azores, the restoration of pastures to native forests would result in an

Table 5.2 Quantitative analysis of the supply of ecosystem services in the Iberian Peninsula, on agricultural land (CLC 2006) (EEA 2013), top 5% high wilderness areas (Carver 2010) and land currently cultivated and projected to be abandoned by 2030 (see Mapping methods section).

				Agricultural Area	**Top 5% high wilderness**	**Projected to be abandoned**	**p value (Kruskall Wallis test)**
Category	**Service**	**Indicator**	**Unit**	**Mean per km^2 (sem)**			
Provisioning	Food production	HANPP	gC/m2/yr	**279.03 (0.21)**	224.12 (2.65)	271.15 (2.91)	2.2E-32 (***)
	Timber	total stock	m^3/há	0.81E+05 (0.21E+03)	2.10 E+05 (1.67E+03)	1.34E+05 (1.68E+03)	0.23 (NS)
	Freshwater	Surface water flow	Mm	151.05 (0.15)	156.08 (0.58)	**244.77 (1.61)**	1.2E-217(***)
Regulation and Maintenance	Climate regulation	Carbon stock	ton/há	24.82 (0.06)	63.19 (0.30)	**74.81 (0.49)**	3.3E-24 (***)
		Net Ecosystem Productivity (NEP)	mg/m^2/year	5.13E+05 (0.34E+03)	6.99 E+05 (1.75E+03)	**7.36E+05 (1.92E+03)**	1.6E-51(***)
	Water regulation	Nitrogen retention	%	2.69 (.00)	**3.04 (.01)**	2.46 (.02)	6.0E-18 (***)
		Soil infiltration capacity	Mm	9.61 (0.02)	15.99 (0.11)	**32.15 (0.23)**	0.0006 (***)
	Air quality	Deposition velocity of Nox	cm/s	0.07 (0.00)	**0.42 (0.00)**	0.28 (0.01)	4.9E-21(***)
Cultural	Recreation	Recreational Potential Index (RPI)	N/A	0.22 (2E-04)	**0.43 (11E-04)**	0.26 (17E-04)	0 (***)

economic benefit of €110 thousand per year from water purification (Cruz and Benedicto 2009). In addition, the benefits of improving water quality upstream impacts lowland areas and human well-being (Forslund et al. 2009). These examples, though limited, demonstrate that protecting and restoring natural vegetation is of economic benefit, and should be integrated under the Water Framework Directive.

Floodplains (wetlands) are also important ecosystems, acting as natural sponges they retain water in river basins, slowly releasing down river and into groundwater. Moreover, they play a fundamental role in filtering out pollutants and are home to protected wildlife. A recent projected scenario revealed that restoring the function of floodplains in EU countries would be saving approximately €1.4 billion of treatment costs for water purification and reduced annual cost of flood damage, currently at €6.4 billion and expected to increase (Feyen and Watkiss 2011). Of course, this type of restoration has initial costs. The Danube Basin restoration project estimates that the recovery of 100,000 ha, would cost 500,000 €/km², i.e. an investment of €500 million. However, this value is still estimated to be much lower than the costs associated to damage control and the improvement of dykes (WWF 2010). Restoration of natural river landscapes has also been estimated to contribute to flood mitigations. In Belgium the Kalkense Meersen has calculated these benefits between €640,000 to €1,654 per annum (Arcadis 2011).

Degradation of natural ecosystems has also been linked to the intensification of natural hazards (Dudley et al. 2010). For example, in the Swiss Alps the protection of old forests contribute to disaster prevention (e.g. avalanches and landslides) and have been analyzed at a value of € 1.6–2.8 billion per year (ISDR 2004). Restoration of natural river landscapes has also been estimated to contribute to flood mitigations. Soil erosion control is another ecosystem service that can potentially benefit economically from rewilding, playing an important role in the prevention and/or mitigation of land degradation and desertification. The role of pristine scrublands against soil erosion was valued at € 44.5/ha, at 2008 throughout Europe and in Belgian grasslands (Ruijgrok and de Groot 2006).

5.5.3. Cultural Benefits

Economic benefits from non-extractive activities such as nature tourism and recreation boost local and regional economies, providing income and employment to communities and private landholders who face limited alternative livelihoods, especially in a context of rural depopulation of marginal areas (Brown and Price 2011; McMorran et al. 2006). Most importantly, the aims of eco-tourism are closely associated with biodiversity conservation. Through the promotion of rewilding efforts, we can speculate that there will be a decrease in fragmented landscapes, creating a break for large mammals and other species (Russo 2006), and indirectly increasing tourism while generating economic benefits to local communities.

Presently, eco-tourism is the fastest growing component sector in tourism (Gössling 2000). Overall, tourism is the largest global economic sector accounting for $3.6trillion in economic activity and eco-tourism has constantly increased 20-30% per year since the early 1990´s (Agrawal and Baranwal 2012) According to the International Ecotourism Society (TIES), eco-tourism involves the responsible visiting to natural areas that conserves the environment and improves the well-being of local people. Wildlife areas appeal to a large spectrum of tourists given the presence of charismatic species and other rare or attractive species. For example the reintroduction of wolves in the Yellowstone National Park has attracted additional tourists, generating economic and social benefits estimated at US$6-9 million per year (Donlan et al. 2006). In Europe, the Pan Parks initiative, a network of wilderness reserves, aims at protecting and managing wilderness areas while promoting sustainable tourism. The reintroduction of ungulates and large carnivores in the Majella and the Retezat National Park in Italy and Romania, respectively, has successfully contributed to the local economy (Kun and van der Donk 2006). In Scotland, tourism from wild landscapes is one of the most important economic sectors, contributing €1.6 billion, annually, to the country´s economy. In particular, recreation opportunities, such as wildlife watching and hillwalking, generate €65 million and support 39,000 full time jobs (Bryden et al. 2010; Brown et al. 2011).

Moreover, the reintroduction of the beaver is thought to potentially generate an additional £2 million per year into the local Scottish economy through eco-tourism (Campbell et al. 2007). In addition to its potential economic benefits, beaver dams are considered to have a positive impact on river systems by increasing both invertebrate and fish populations (Kemp et al. 2010). Although many remain uncertain to beavers positive impacts on nature, quantitative evidence has revealed that the reintroduction of the species results in an increased habitat diversity and abundance of fish, specifically salmon (Kemp et al. 2010; BSWG 2013).

Safeguarding the Romanian Carpathians Ecological Network is a success case study on rewilding initiatives improving the quality of life of those who live there through the development of local economies while focusing on the conservation of natural values and cultural heritage (Maanen et al., 2006). The Carpathian Mountains and the Danube Delta are considered the biodiversity and wilderness hotspots of Romania. In Zarnesti a small community in Romania increased their total local revenue from € 140,000 in 2001 to €

260,000 in 2002 through eco-tourism programmes (Carpathian Large Carnivore Project 2000) The Natura 2000 network further exemplifies another cost effective mean of protecting wildlife while generating benefits. Annually, the gross socio-economic and co-benefits (social and environmental) from the Natura 2000 network range between €223 billion and €314 billion, representing between 2 and 3% of EU´s GDP (ten Brink et al. 2011). This figure contrasts with the annual investment in the Natura 2000 network, estimated at €5.8 billion (Gantioler et al. 2010) while providing 8 million (FTE) jobs (Gantioler et al. 2010). In other words, rewilding could locally develop a new economy around the use of wilderness through the creation of new markets, such as eco-tourism.

5.6. Discussion

The degradation, or land conversion, of natural ecosystems alters not only species richness and composition; it reduces ecosystem functionality, impacting the flow of ecosystem services, the costs of recuperation and ultimately human well-being (Flynn et al. 2009). Agriculture in Europe has taken two different paths, the marginalization of agriculture in rural mountain landscapes and the intensification in regions with more fertile soils (MacDonald et al. 2000; Strijker 2005). The abandonment of extensive agriculture, mainly in mountain areas, has been a result of various drivers leading to rural exodus. Years of combating rural desertification and the maintenance of agricultural practices through incentives has not contributed to the attenuation of this phenomenon (Merckx and Pereira, in review).

The management of these abandoned lands has become a challenge for many conservationists, as policies and government strategies have recently integrated the restoration of ecosystem services alongside the conservation of biodiversity. According to the CBD Global Biodiversity Outlook for 2010 the opportunity for restoring nature across Europe through rewilding is now (SCBD 2010). The restoration of nature through rewilding is seen as a solution to the on-going land abandonment, developing a bold new economy while offering social and environmental benefits, based on the restoration and sustainability of ecosystem services provided by wildlife, wild areas and wilderness (McMorran et al. 2006; Hein 2011; Gantioler et al. 2010; Donlan et al. 2006; Bryden et al. 2010; Brown et al. 2011). A recent study has also determined a cost-benefit analysis of restoration projects and determined that ecological restoration results in positive investments (de Groot et al. 2013). In this analysis, we investigated the existence of a spatial co-occurrence of a gradient of both wilderness and ecosystem services supply (Fig. 5.2). In addition, we looked into the impacts that rewilding efforts post land abandonment could potentially have on the supply of ecosystem services. Overall, we found positive indicators that high degrees of wilderness co-occur with a high supply of ecosystem services (Fig. 5.2.a). This spatial co-occurrence appears even stronger when looking into regulating and cultural services (Fig 5.2c and 5.2d). Furthermore, our results provide quantitative evidence that the opportunity of restoring abandoned land to a self-sustained natural state (rewilding) could increase the supply of regulating and cultural services (Table 5.2). We thus argue that by restoring and sustaining wilderness areas we are underpinning a supply of high quality ecosystem services provided by wild areas. In other words rewilding can be used as a conservation management tool and target specific ecosystem services. For example, the expansion of wilderness areas will play an important role in mitigating climate change through the sequestration of carbon. These services will also heighten a new economy market, based on services supplied by wilderness, providing an economic break for viable rural communities through the creation of jobs and income generated from incentives (e.g. PES, carbon markets and eco-tourism). However, the implementation of incentives in remote areas faces major limitations when land abandonment has already occurred and we no longer have inhabitants.

Although, the concept of rewilding is fairly recent in Europe, it has been identified as a cost-effective management strategy for traditional land uses in Scotland (McMorran et al. 2006; Brown et al. 2011) In the Netherlands, the promotion of rewilding has been positively perceived. Individuals attribute a low willingness to pay for the conservation of extensive farming versus rewilding initiatives, which were generally ranked high in terms of attractiveness (van Berkel and Verbug 2012). However, we cannot generalize rewilding as the only cost-effective strategy, positively perceived by all rural inhabitants.

From a holistic perspective, both active and passive restoration promoting wilderness needs to be viewed as a management strategy that provides both benefits to humans through the flow of ecosystem services, while preserving biodiversity. Nonetheless, the application of these two management strategies is dependent on the state of the ecosystem (Chazdon 2008), the climatic, biophysical, socio-economic elements of the location, as well as the costs and benefits associated to each management option. Thus, the total eradication of human intervention needs to be pondered. Semi-natural grasslands rely on adequate management regimes, through grazing activity. In Europe, decreases in pasturing activity, has lead to the natural encroachment of vegetation, reducing landscape heterogeneity, impacting those species associated to open spaces and as a result the loss of functional groups (Laiolo et al. 2004;Lindborg and Eriksson 2004; Peco et al. 2012).

Consequently, protecting these ecosystems involves the reinstatement of natural disturbances regimes, safeguarding ecosystem processes. Another limitation with rewilding is the time to restoration, depending on the type and the degree of intervention that is implemented. Natural forest regeneration in Europe, for example can take from 20 years to nearly a century (Verburg and Overmars 2009), depending on the cultivation history and on the geographical conditions. In these cases, the return on investments can be long, before the supply of ecosystem services starts increasing.

We are not suggesting that rewilding efforts through active or passive restoration be the only solution to Europe´s present situation, but be considered as a potential strategy in those areas where the social-ecological dynamics of the landscape are no longer socially, economically and environmentally sustainable. Yet, there are still many challenges in understanding the full relationship between landscape management and the supply of ecosystem services and the economic benefits and costs associated to each management type and ideally how they can be framed into wilderness areas and adopted in environmental policies.

There are still many questions to be answered concerning rewilding efforts throughout Europe. However, we argue that the promotion of the restoration of those regions undergoing abandonment is an opportunity for ecosystems services and biodiversity. By protecting and increasing wilderness we are underpinning areas of high ecosystem service. Notwithstanding, several pitfalls and trade-offs can be associated to rewilding. We have yet to determine how the promotion of rewilding would affect social-ecological systems, primarily humans' adaptive capacity to changes in the provision of ecosystem services. Another consequence to rewilding is the potential loss of traditional cultural values and heritage and its social, environmental implications are still unknown (Cerqueira et al. 2010).

The emerging balance calls for further research and increase awareness of the environmental, social and economic benefits associated to wilderness areas. Raising awareness of these benefits may help to promote the concept and reinforce the idea of how naturalness is an opportunity for increasing overall human well-being and defining public policies and funding of nature conservation policies.

5.7. References

Arcadis. (2011). Recognizing Natura 2000 Benefits and Demonstrating the Economic Benefits of Conservation Measures. Final report. Development of a Tool for

ValuingConservation Measures. Project number 11639 | version 2 | 31-10-2011.

Benayas, J. M. R., Newton, A. C., Diaz, A., & Bullock, J. M. (2009). Enhancement of Biodiversity and Ecosystem Services by Ecological Restoration: A Meta-Analysis. *Science*, *325*(5944), 1121–1124.

BSWG. (2013). Beaver-salmonid interactions in Scotland: Scoping the immediate action required towards production of the beaver-salmonid reportto the EnvironmenMinister.Workshop Proceedings July 3rd.

Bishop, J. & Timberlake, L. (2007). Business and Ecosystems: Markets for Ecosystem Services-New Challenges and Opportunities for Business and the Environment. 20pp.

BirdLife International. (2011). Rewilding may offer a sustainable alternative to traditional management. Presented as part of the BirdLife State of the world's birds website. Available from: http://www.birdlife.org/datazone/sowb/casestudy/415. Checked: 01/10/2013

Brown, C., Mcmorran, R., & Price, M. F. (2011). Rewilding – A New Paradigm for Nature Conservation in Scotland? *Scottish Geographical Journal*, *127*(4), 288–314. doi:10.1080/14702541.2012.666261

Bryden, D. M., Westbrook, S. R., Burns, B., Taylor, W. A. & Anderson, S. (2010). Assessing the economic impacts of nature based tourism in Scotland *Scottish Natural Heritage Commissioned Report No*. 398.

Campbell, R. D., Dutton, A. & Hughes, J. (2007). Economic impacts of the beaver. Report for the Wild Britain Initiative. 28 pages.

Carpathian Large Carnivore Project (2000) Annual Report, Zarnesti Romania (www.clcp.ro).

Carver, S. (2010). Chapter 10.3 Mountains and wilderness, in European Environment Agency Europe´s ecological backbone: Recognizing the true value of our mountains, EEA Report No 6/2010, pp. 192-201.

Cerqueira, Y., Araújo, C., Vicente, J., Pereira, H., & Honrado, J. (2010). Ecological and Cultural Consequences of Agricultural Abandonment in the Peneda-Gerês National Park (Portugal). In N. Evelpidou, T. Figueiredo, F. Mauro, V. Tecim, & A. Vassilopoulos (Eds.), *Natural Heritage from East to West* (pp. 175–183). Springer Berlin Heidelberg. Retrieved from http://dx.doi.org/10.1007/978-3-642-01577-9_22

Chan, K. M. A., Shaw, M. R., Cameron, D. R., Underwood, E. C., & Daily, G. C. (2006). Conservation Planning for Ecosystem Services. *PLoS Biol*, *4*(11), e379. doi:10.1371/journal.pbio.0040379

Costanza, R., & Mageau, M. (1999). What is a healthy ecosystem? *Aquatic Ecology*, *33*(1), 105–115. doi:10.1023/A:1009930313242

Cruz, A. de la & Benedicto, J. (2009). Assessing Socio-economic Benefits of Natura 2000-a Case Study on the ecosystem services provided by SPA PICO DA VARA/RIBEIRO DO GUILHERME. Output of the Project Financing Natura 2000: Cost estimate and benefits of Natura 2000 (Contract No.:070307/2007/484403/MAR/B2). 43pp.

Dax, T. (2005). The redefinition of Europe's less favoured areas. In: Rural development in Europe – 3rd annual conference – Funding European Rural Development in 2007–2013. MPRA paper no. 711.

De Groot, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., … van Beukering, P. (2012). Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem Services*, *1*(1), 50–61. doi:10.1016/j.ecoser.2012.07.005

De Groot, R. S., Blignaut, J., van der Ploeg, S., Aronson, J., Elmqvist, T., & Farley, J. (2013). Benefits of Investing in Ecosystem Restoration. *Conservation Biology*, *27*(6), 1286–1293. doi:10.1111/cobi.12158

Deng, L., Shangguan, Z.-P., & Sweeney, S. (2013). Changes in Soil Carbon and Nitrogen following Land Abandonment of Farmland on the Loess Plateau, China. *PLoS ONE*, *8*(8), e71923. doi:10.1371/journal.pone.0071923

Donlan, J., Berger, J., Bock, C. E., Bock, J., Burney, D. A., Estes, J. A., Foreman, D.,Martin, P., Roemer, G., Smith, F., Soule, M., & Greene, H. (2006). Pleistocene Rewilding: An optimistic agenda for Twenty First Century conservation. The American Naturalist, 168: 660-681.

Dudley, N., S. Stolton, A. Belokurov, L. Krueger, N. Lopoukhine, K. MacKinnon, T. Sandwith and N. Sekhran [editors] (2010); *Natural Solutions: Protected areas helping people cope with climate change*, IUCNWCPA, TNC, UNDP, WCS, The World Bank and WWF, Gland, Switzerland, Washington DC and New York, USA.

Dunbar, M. B., Panagos, P., & Montanarella, L. (2013). European perspective of ecosystem services and related policies. *Integrated Environmental Assessment and Management*, *9*(2), 231–236. doi:10.1002/ieam.1400

Egoh, B., Reyers, B., Rouget, M., Richardson, D. M., Le Maitre, D. C., & van Jaarsveld, A. S. (2008). Mapping ecosystem services for planning and management. *Agriculture, Ecosystems & Environment*, *127*(1–2), 135–140. doi:10.1016/j.agee.2008.03.013

European Commission (2011a). Our life insurance, our natural capital: an EU biodiversity strategy to 2020. Brussels, 2011-05-03. COM(2011) 244.

European Commission (2011b). Regional policy contributing to sustainable growth in Europe 2020. COM(2011)17 final. Brussels.

European Commission (2008): Review of the Less Favoured Area Scheme. Available et http://ec.europa.eu [Quoted 7.10.2008]

EEA. (2010). *Integrated assessment of Europe's mountain areas*. EEA, Copenhagen. Forthcoming September 2010.

EEA. (2006). Progress towards Halting the Loss of Biodiversity by 2010. EEA Report No. 5/2006. Luxembourg: European Union Directorate-General Environment, Nuclear Safety and Civil Protection. View[accessed on 18 July 2008].

EEA. (2013). Corine Land Cover data base 2006

Ewert, A., Overholt, J., Voight, A., Wang, C. (2011). Understanding the Transformative Aspects of the Wilderness and Protected Lands Experience upon Human Health. USDA Forest Service Proceedings RMRS-P-64.

Ferraz, P., Luz, D. A., Azevedo, T., … Metzger, J. (2014). How good are tropical forest patches for ecosystem services provisioning? *Landscape Ecology*, *29*(2), 187–200. doi:10.1007/s10980-014-9988-z

Feyen, L. & Watkiss, P. (2011). Technical Policy Briefing Note 3. The Impacts and Economic Costs of River Floods in Europe, and the Costs and Benefits of Adaptation. Results from the EC RTD Climate Cost Project. In Watkiss, P (Editor), 2011. The Climate Cost Project. Final Report. Published by the StockholmEnvironment Institute, Sweden, 2011. ISBN. 978-91-86125-35-6.

Flynn, D. F. B., Gogol-Prokurat, M., Nogeire, T., Molinari, N., Richers, B. T., Lin, B. B., DeClerck, F. (2009). Loss of functional diversity under land use intensification across multiple taxa. *Ecology Letters*, *12*(1), 22–33. doi:10.1111/j.1461-0248.2008.01255.x

Forslund, A., et al. (2009). Securing Water for Ecosystems and Human Well-being: The Importance of Environmental Flows. Swedish Water House Report 24. SIWI.

Gantioler, S., Rayment, M., Bassi, S., Kettunen, M., McConville, A., Landgrebe, R., Gerdes, H. & ten Brink, P. (2010) *Costs and Socio-Economic Benefits associated with the Natura 2000 Network.* Final report to the European Commission, DG Environment on Contract ENV.B.2/SER/2008/0038. Institute for European Environmental Policy / GHK / Ecologic,Brussels 2010

Gössling, S. (2000). Tourism – sustainable development option? *Environmental*

Conservation, *27*(03), 223–224.

Grêt-Regamey, A., Rabe, S.-E., Crespo, R., Lautenbach, S., Ryffel, A., & Schlup, B. (2014). On the importance of non-linear relationships between landscape patterns and the sustainable provision of ecosystem services. *Landscape Ecology*, *29*(2), 201–212. doi:10.1007/s10980-013-9957-y

Haberl, H., Erb, K. H., Krausmann, F., Gaube, V., Bondeau, A., Plutzar, C., … Fischer-Kowalski, M. (2007). Quantifying and mapping the human appropriation of net primary production in earth's terrestrial ecosystems. *Proceedings of the National Academy of Sciences*, *104*(31), 12942–12947.

Haines-Young, R., & Potschin, M. (2013). Common International Classification of Ecosystem Services (CICES): Consultation on Version 4, August-December 2012. *EEA Framework Contract No* EEA/IEA/09/003

Harrison, P., Vandewalle, M., Sykes, M., Berry, P., Bugter, R., Bello, F., … Zobel, M. (2010). Identifying and prioritising services in European terrestrial and freshwater ecosystems. *Biodiversity and Conservation*, *19*(10), 2791–2821. doi:10.1007/s10531-010-9789-x

Hein, L. (2011). Economic Benefits Generated by Protected Areas: The Case of the Hoge Veluwe Forest, the Netherlands. *Ecology and Society*, *16*. Retrieved from http://hdl.handle.net/10535/7603

Heintzman, P. (2013). Spiritual Outcomes of Park Experience:A Synthesis of Recent Social Science Research. The George Wright Forum, 30(3):273-279.

Hill, N. R. (2007). Wilderness therapy as a treatment modality for at-risk youth: a primer for mental health counselors. Journal of Mental Health Counseling, 29(4)

Huhtala, M., Kajala, L. & Vatanen, E. (2010). Local economic impacts of national park visitors spending in Finland: The development process of an estimation method. Metlan tyoraportteja / Working papers of the Finnish Forest Research Institute 149. Available at:http://www.metla.fi/julkaisut/workingpapers/2010/index_en.htm

ICPDR.(2010). Danube catch-the magazine of the Danube River/www.ICPR.ORG 3-4/2010.

Kemp, P.S., Worthington, T.A. & Langford, T.E.L. (2010). A critical review of the effects of beavers upon fish and fish stocks. Scottish Natural Heritage Commissioned Report No. 349 (iBids No. 8770).

Kenny, A. (2006). Ecosystem Services in the New York City Watershed.The Ecosystem Market Place.

Kettunen, M., Vihervaara, P., Kinnunen, S., D'Amato, D., Badura, T., Argimon, M. and Ten Brink, P. (2012). Socio-economic importance of ecosystem services in the Nordic Countries. Synthesis in the context of The Economics of Ecosystems and Biodiversity (TEEB).

Klein Goldewijk, K., Beusen, A., van Drecht, G., & de Vos, M. (2011). The HYDE 3.1 spatially explicit database of human-induced global land-use change over the past 12,000 years. *Global Ecology and Biogeography*, *20*(1), 73–86. doi:10.1111/j.1466-8238.2010.00587.x

Körner, C. and M. Ohsawa (coordinating lead authors), et al. 2005. Mountain Systems. Chapter 24. In: R. Hassan, R. Scholes, and N. Ash, (eds.). Ecosystems and human well-being: current state and trends, Volume 1. Washington, DC: Island Press. p. 681-716.

Körner, C. (2002). "Mountain biodiversity, its causes and function: an overview". In: Korner, C. & Spehn, E. (eds), *Mountain biodiversity: a global assessment*. Parthenon, New York, pp. 3-20.

Kremen, C., Williams, N. M., & Thorp, R. W. (2002). Crop pollination from native bees at risk from agricultural intensification. *Proceedings of the National Academy of Sciences*, *99*(26), 16812–16816.

Kun, Z., & van der Donk, M. (2006). Providing wilderness experience opportunities in Europe's certified PAN parks. PARKS-The visitor experience challenge, 16(2):34-40.

Laiolo, P., Dondero, F., Ciliento, E., & Rolando, A. (2004). Consequences of pastoral abandonment for the structure and diversity of the alpine avifauna. *Journal of Applied Ecology*, *41*(2), 294–304. doi:10.1111/j.0021-8901.2004.00893.x

Lindborg, R., & Eriksson, O. (2004). Effects of Restoration on Plant Species Richness and Composition in Scandinavian Semi-Natural Grasslands. *Restoration Ecology*, *12*(3), 318–326. doi:10.1111/j.1061-2971.2004.00334.x

Maanen, E.V., Predoiu, G., Klaver, R., Soulé, M., Popa, M., Ionesco, O., Jurj, R., Negus, S., Ionescu, G. & Altenburg, W. 2006. Safeguarding the Romanian Carpathian Ecological Network. A vision for large carnivores and biodiversity in Eastern Europe. A&W ecological consultants, Veenwouden, The Netherlands. Icas Wildlife Unit, Brasov, Romania.

MacDonald, D., Crabtree, J. ., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., … Gibon, A. (2000). Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. *Journal of Environmental Management*, *59*(1), 47–69. doi:10.1006/jema.1999.0335

Maes, J., Teller, A., Erhard, M., Liquete, C., Braat, L., Berry, P., Egoh, B., Puydarrieux, P., Fiorina, C., Santos, F., Paracchini, M. L., Keune, H., Wittmer, H., Hauck, J., Fiala, I., Verburg, P.H., Condé, S., Schägner, J.P., San Miguel, J., Estreguil, C,, Ostermann, O., Barredo, J.I., Pereira, H.M., Stott, A., Laporte, V., Meiner, A., Olah, B., Royo Gelabert, E., Spyropoulou, R., Petersen, J.E., Maguire, C., Zal, N., Achilleos, E., Rubin, A., Ledoux, L., Brown, C., Raes, C., Jacobs, S., Vandewalle, M., Connor, D., Bidoglio, G. (2013). Mapping and assessment of Ecosystems and their Services. An analytical framework for ecosystem assessments under action 5 of the EU biodiversity strategy to 2020. Publications office of the European Union, Luxembourg.

Maes, J., Paracchini, M.L., & Zulian, G. (2011). A European Assessment of the Provision of Ecosystem Services: Towards an Atlas of Ecosystem Services. Publications Office of the European Union, Luxembourg, doi:10.2788/63557, p. 81.

Maes, J., Paracchini, M. L., Zulian, G., Dunbar, M. B., & Alkemade, R. (2012a). Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. *Biological Conservation*, *155*(0), 1–12. doi:10.1016/j.biocon.2012.06.016

Maes, W. H., Heuvelmans, G., & Muys, B. (2009). Assessment of Land Use Impact on Water-Related Ecosystem Services Capturing the Integrated Terrestrial−Aquatic System. *Environmental Science & Technology*, *43*(19), 7324–7330. doi:10.1021/es900613w

Maes, J., Hauck, J., Paracchini, M. L., Ratamäki, O., Termanse, M., Perez-Sboa, M., Kopperoinen, L., Rankinen, K., Schägner, J. P., Henrys, P., Cisowska, I., Zandersen, M., Jax, K., La Notte, A., Leikola, N., Pouta, E., Smart, S., Hasler, B., Lankia, T., Andersen, H. E., Lavalle, C., Vermaas, T., Alemu, M. H., Scholefield, P.,Batista, F., Pywell, R., Hutchins, M., Blemmer, M., Fonnesbech-Wulff, A., Vanbergen, A. J., Münier, B., Baranzelli, C., Roy, D., Thieu, V., Zulian, G., Kuussaari, M., Thodsen, H., Alanen, E.L., Egoh, B., Sørensen, P. B., Braat, L. & Bidoglio, G. (2012b). *A spatial assessment of ecosystem services in Europe: methods, case studies and policy analysis. - phase 2.* PEER Report No 4. Ispra: Partnership for European EnvironmentalResearch

MA (Millennium Ecosystem Assessment) 2005. Ecosystem and human well-being. Synthesis. Island Press. Washington D. C. 137 p.

McMorran, R., Price, M. F. & McVittie, A. (2006). A review of the benefits and opportunities attributed to Scotland´s landscapes of wild character. Scottish Natural Heritage Commissioned Report No. 194 (ROAME No. F04NC18).

Merckx, T. & Pereira, H. M. (submitted). Reshaping agri-environmental subsidies: From marginal farming to large-scale rewilding. Basic and Applied Ecology.

Naidoo, R., & Ricketts, T. H. (2006). Mapping the Economic Costs and Benefits of Conservation. *PLoS Biol*, *4*(11), e360. doi:10.1371/journal.pbio.0040360

Natural England Commissioned Report, 2012. NECR101. Valuing Ecosystem Services: Case Studies from Lowland England. Annex 4- Knepp Castle Estate Re-lding:Sussex. Published 15 August 2012.

Nelson, E., Sander, H., Hawthorne, P., Conte, M., Ennaanay, D., Wolny, S., … Polasky, S. (2010). Projecting Global Land-Use Change and Its Effect on Ecosystem Service Provision and Biodiversity with Simple Models. *PLoS ONE*, *5*(12), e14327. doi:10.1371/journal.pone.0014327

Peco, B., Carmona, C. P., de Pablos, I., & Azcárate, F. M. (2012). Effects of grazing abandonment on functional and taxonomic diversity of Mediterranean grasslands. *Agriculture, Ecosystems & Environment*, *152*(0), 27–32. doi:10.1016/j.agee.2012.02.009

Pereira, E., Queiroz, C., Pereira, H. M., & Vicente, L. (2005). Ecosystem Services and Human Well-Being: a Participatory Study in a Mountain Community in Portugal. *Ecology and Society*, *10*(2). Retrieved from http://www.ecologyandsociety.org/vol10/iss2/art14/

Petter, M., Mooney, S., Maynard, S. M., Davidson, A., Cox, M., & Horosak, I. (2013). A Methodology to Map Ecosystem Functions to Support Ecosystem Services Assessments. *Ecology and Society*, *18*(1). doi:10.5751/ES-05260-180131

Price, M.F., Moss, L.A.G. & Williams, P.W. (1997). Tourism and amenity migration. *In* B. Messerli & J.D. Ives, eds. *Mountains of the world: a global priority,* pp. 249-280. New York, USA and Carnforth, UK, Parthenon Publishing Group.

Raudsepp-Hearne, C., Peterson, G. D., & Bennett, E. M. (2010). Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proceedings of the National Academy of Sciences*. Retrieved from http://www.pnas.org/content/early/2010/02/24/0907284107.abstract

Reed, M. S., Hubacek, K., Bonn, A., Burt, T. P., Holden, J., Stringer, L. C., … Worrall, F. (2013). Anticipating and Managing Future Trade-offs and Complementarities between Ecosystem Services. *Ecology and Society*, *18*(1). doi:10.5751/ES-04924-

180105

Ruijgrok E.C.M. & de Groot, R.S. de (2006). Kentallen Waardering Natuur, Water, Bodem En Landschap. Hulpmiddel Bij Mkba's.

Russo, D. (2006). Effects of land abandonment on animal species in Europe: conservation and management implications. Integrated assessment of vulnerable ecosystems under global change in the EU. Project report. 52 pp.

Schils, R., Kuikman, P., Liski, J., van Oijen, M., Smith, P., Webb, J., alm, J., Somogyi, Z., van den Akker, J., Billett, M., Emmett, B., Evans, c., Lindner, M., Palosuo, T.,Bellamy, P., Alm, J., Jandl, R., Hiederer, R. (2008). Review of existing information on the interrelations between soil and climate change. Technical Report - 2008 – 048.

Schmitt, M., Bahn, M., Wohlfahrt, G., Tappeiner, U., & Cernusca, A. (2010). Land use affects the net ecosystem CO2 exchange and its components in mountain grasslands. *Biogeosciences*, *7*(8), 2297–2309. doi:10.5194/bg-7-2297-2010

Schrötter, D., Cramer, W., Leemans, R., Prentice, I. C., Araújo, M. B.,Arnell, N. W., Bondeau, A., Bugmann, H., Carter, T. R., Gracia, C., Vega-Leinert, A. C., Erhard, M., Ewert, F., Glendining, M., House, J. I., Kankaanpaa, S., Klein, R. J. T et al. (2005). Ecosystem Service Supply and Vulnerability to Global Change in Europe. Science, 310, 1333-1337.

SCBD - Secretariat of the Convention on Biological Diversity. (2010). Global Biodiversity Outlook 3. Montréal, 94 pages.

Silva, J.P., Toland, J., Jones, W., Eldridge, J., Thorpe, E. & O'Hara, E. (2008). LIFE and Europe's Grasslands. Restoring a Forgotten Habitat.LIFE Focus series, Luxembourg: Office for Official Publications of the European Communities.

Strijker, D. (2005). Marginal lands in Europe–causes of decline. Basic Applied Ecology, 6, 99–106.

Tallis, H., Kareiva, P., Marvier, M., & Chang, A. (2008). An ecosystem services framework to support both practical conservation and economic development. *Proceedings of the National Academy of Sciences*, *105*(28), 9457–9464.

TEEB (The Economics of Ecosystems and Biodiversity). 2010. The economics of ecosystems and biodiversity—ecological and economic foundations. P. Kumar, editor. Earthscan, London.

TEEB (2012). The Economics of Ecosystems and Biodiversity in Local and Regional Policy and Management. Edited by Heidi Wittmer and Haripriya Gundimeda. Earthscan, London and Washington.

Ten Brink, P., Badura, T., Bassi, S., Daly, E., Dickie, I., Ding, H., Gantioler, S., Gerdes, H., Hart, K., Kettunen, M., Lago, M., Lan,S., Markandya, A., Mazza, L., Nunes, P.A.L.D., Pieterse, M., Rayment, M. & Tinch, R. (2011). Estimating the Overall Economic Value of the Benefits provided by the Natura 2000 Network. Final Synthesis Report to the European Commission, DG Environmental on Contract ENV.B.2/SER/2008/0038. Institute for European Environmental Policy/GHK/Ecologic, Brussels 2011

Van Berkel, D. B., & Verburg, P. H. (2014). Spatial quantification and valuation of cultural ecosystem services in an agricultural landscape. *Ecological Indicators*, *37, Part A*(0), 163–174. doi:10.1016/j.ecolind.2012.06.025

Vanacker, V., Bellin, N., Molina, A., & Kubik, P. (2014). Erosion regulation as a function of human disturbances to vegetation cover: a conceptual model. *Landscape Ecology*, *29*(2), 293–309. doi:10.1007/s10980-013-9956-z

Väre, H., Lampinen, R., Humphries, C., & Williams, P. (2003). Taxonomic Diversity of Vascular Plants in the European Alpine Areas. In L. Nagy, G. Grabherr, C. Körner, & D. A. Thompson (Eds.), *Alpine Biodiversity in Europe* (Vol. 167, pp. 133–148). Springer Berlin Heidelberg. Retrieved from http://dx.doi.org/10.1007/978-3-642-18967-8_5

Verburg, P., & Overmars, K. (2009). Combining top-down and bottom-up dynamics in land use modeling: exploring the future of abandoned farmlands in Europe with the Dyna-CLUE model. *Landscape Ecology*, *24*(9), 1167–1181. doi:10.1007/s10980-009-9355-7

Viviroli, D., Dürr, H. H., Messerli, B., Meybeck, M., & Weingartner, R. (2007). Mountains of the world, water towers for humanity: Typology, mapping, and global significance. *Water Resources Research*, *43*(7), W07447. doi:10.1029/2006WR005653

Wild Europe. (2012). A Working Definition of European Wilderness and Wild Areas. Discussion Draft. http://www.panparks.org/sites/default/files/docs/iyw/Definition-of-wilderness-for-Europe.pdf

WWF. 2010. Assessment of the restoration of potential along the Danube and main tributaries. Final Draft, Vienna, May 2010.

Chapter 6. Discussion, Synthesis and Conclusions

6.1. Summary of main results and key findings

The main results and key findings from the studies described in the previous chapters can be summarized as follows:

- pressures and frictions of agro-pastoral abandonment vary temporally and spatially, presently and in the future; CAP subsidies are the main friction holding back abandonment; local people's willingness to leave seems to be strongly related with their livestock numbers;

- human preference for landscape and its elements, as well as local perception of landscape change, varied along an urban-rural gradient; the protection of ecosystem services and the local economic development through tourism were identified as primary strategies for a mountainous region of Portugal undergoing abandonment; landscape planers can use this better understanding of local stakeholders' preference for ecosystems and their services, and of how their perceptions are reflected on the landscape;

- when analyzing the temporal dynamics of ecosystem services provision at the local level, we determined an increase in carbon storage/sequestration and soil conservation with land abandonment; we also predicted a decrease in local richness of those plant species with a high affinity to agricultural systems, but no apparent impact on other tested groups;

- the supply of ecosystem services varies along an wilderness quality gradient across Europe, with high ecosystem service supply associated to mountain systems, especially in the case of regulating and cultural services; in Europe the promotion of future rewilding in those areas presently associated with agro-ecosystems which have gone through abandonment, or potentially facing abandonment, would thus result in an increased supply of regulating and cultural services.

These and other results are discussed from an integrative perspective in the following sections.

6.2 Land abandonment and its drivers

Throughout centuries, rural mountain inhabitants developed agricultural and silvi-pastoral techniques which permitted the exploitation of natural resources in an apparently sustainable way. However, particularly since the middle of the 20th century, socioeconomic and technologic development of human societies has been driving profound shifts in the classic relationships between cities and the countryside (Gutman 2007), ultimately inducing changes in rural landscapes.

The abandonment of agricultural land is a growing concern throughout much of Europe's rural mountain areas. Migration and aging population are the main reasons behind the collapse of traditional farming systems and the increase in land abandonment (Rey Benayas 2007; see chapter 2). Together with an intrinsic resistance to adopting modern market oriented farming practices, these processes induce consequences (still under-evaluated) to the environment as well as numerous socioeconomic impacts (Benayas et al. 2007).

Many of the most traditional types of agricultural landscapes in Europe are dramatically decreasing due to this partial or complete abandonment of farming (EEA 2005). Some of the most critical nature conservation issues today relate to changes in traditional farming practices on habitats such as hay meadows, lowland wet grasslands, dry grasslands and arable land (Henle et al. 2008; Halada et al. 2011). Overall, these habitats usually disappear after the abandonment of traditional farming practices, and species adapted to the diversity of structures or resources in such High Nature Value (HNV) farmlands may not survive (Henle et al. 2008). Consequently these simplified, homogenous landscapes resulting from agricultural abandonment often have reduced biodiversity when compared to the more diverse HNV farmland areas (Henle et al. 2008).

The identification of the drivers responsible for these changes has become imperative. However, the various drivers are complex as they fluctuate through time and space and are locally, regionally and even globally induced (Bürgi et al., 2004). Additionally, our findings further suggest that, throughout time, some drivers change from a promoter to an inhibitor of land abandonment (see Chapter 2). According to our results and those of other recent studies, the present drivers are expected to continue in the near future applying further pressure and consequently leading to augmentation of agro-pastoral abandonment (Keenlyside and Tucker 2010).

On the other hand, the number of pressures has increased along the years. In the past, economic reasons were the main pressures behind rural exodus, today these communities are faced with further challenges, such as limited social amenities. These include the closing of services such as schools and health centers (Pereira et al. 2005). This driver not only leads to the out migration of residents but also acts as an inhibitor to new residents. Yet, there are those drivers which act as inhibitors; these consist of mainly social and economic motives. The CAP subsidies have been identified as the primary friction holding back abandonment, which suggests that any future cuts to Pillar 2 could potentially mitigate future agro-pastoral activities (Keenlyside and Tucker 2010). Our findings highlight that, those individuals with greater livestock numbers also receive higher subsidies and were also indentified as those who were less willing to leave the rural community.

These findings suggest that if farmland utility is high, or in other words, if there is high economic return, then individuals are less likely to migrate (Figueiredo and Pereira 2012). However, if migration was to occur in the presence of high farmland utility, we may attribute this to potential social-bonds. Determining social thresholds associated to land abandonment provides landscape planners and policy makers with probable social dynamic scenarios to various drivers. According to the social-ecological model of farmland abandonment proposed by Figueiredo and Pereira (2012), migration is a collective behavior that is both socially and economically driven. Our results recognized economic reasons as the main driver of abandonment and rural exodus, and we pin-pointed social bond as the main factor of changes in residency within the parish (i.e. from one village to another).

Presently, the pressures to abandon agricultural activities outweigh the frictions against this trend (Beilin et al. 2014). Therefore European policies need to consider migration as an influential factor on land use change and not consider these as disconnected issues. Besides the various driving forces resulting in land abandonment, local landscape planners need to consider how present residents use their surroundings. By determining local preference and value for a particular ecosystem we are identifying human-environmental interactions. According to the conceptual model of human-environmental interactions proposed by Gobster et al. (2007), ecological aesthetics affects landscape planning, design and management. Many of the empirical studies performed before have been focused on assessing only aesthetic preferences (Hunziker et al. 2008; Van den Berg and Koole 2006; Kaltenborn and Bjerke 2002), which have unveiled to play a key part in driving land use change (Gobster et al. 2007; Hunziker et al. 2008).

Documenting human-landscape relationship is complex but, as already stated, fundamental in management policies and specifically within conservation projects supporting goals at the local scale (Hunziker et al., 2008; Natori and Chenoweth, 2008;

Sayadi et al., 2009; Soliva and Hunziker, 2009). Our findings provide a first analysis of human preference for various ecosystems and services along an urban-rural gradient (see Chapter 3). Additionally, locals identified those ecosystems of conservation and aesthetic preference which allowed us to make a connection between the two. The top two ecosystems (i.e. forests and terraced land) were of both conservation and aesthetic value. This supports Gobster et al.'s (2007) model that landscapes of aesthetic appreciation are normally those regions in which humans feel a need to protect. Establishing not only aesthetic preference for different landscapes, but the human preference for ecosystems and their services along a rurality gradient, also contributes to the on-going need to recognize the spatial demand of a particular service and plan for multi-functional landscapes (Soliva and Hunziker 2009; Surová and Pinto-Correia 2008).

The abandonment of traditional agricultural landscapes is also thought to have potential impacts on traditional ecological knowledge (TEK). The loss of TEK has become a growing concern throughout Europe in the last decade, mainly due to its potential contribution in providing information for resilience building (Plieninger and Bieling 2013) as well as for environmental policy and management (Kimmerer 2012). The erosion of TEK has been associated to several factors, but current literature has identified industrialization of land use and shifts as well as the transition to a market economy as the main determinants (Gómez-Baggethun et al. 2010; Glasenapp and Thornton 2011). The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) has recognized the potential TEK may have when complemented with scientific knowledge to support policy makers. Nonetheless, its role in land management throughout Europe has only recently come into play (Carvalho and Frazão-Moreira 2011). The existence of different preferences for a given natural resource, ecosystem or ecosystem service by different stakeholder groups or individuals, and the role they play in managing these resources through TEK, is indispensable and the basis for a successful management (Bassi and Tache, 2011; Emery et al. 2014).

6.3 Consequences of abandonment for biodiversity and ecosystem services

The abandonment of agricultural land throughout much of Europe's rural mountain areas has induced shifts in the structure and properties of ecosystems and landscapes, which in turn control the condition and trends of ecosystem services and biodiversity (MA 2005; Pereira et al. 2005; Metzger et al. 2006). Much of Europe´s mountain areas which have not been abandoned are characterized by low intensity farming systems, linked to the preservation of biodiversity and important ecosystems (Bignal and McCracken 1996, Moreira et al. 2001, Metzger et al. 2006, Reidsma et al. 2006). The preservation of semi-natural biotopes and landscapes such as pasturelands is desirable and important for biodiversity.

The strong international concern regarding the consequences of agricultural abandonment results from the recognition that the most important anthropogenic cause of agro-ecosystem biodiversity loss is rapid land use and land cover change and the subsequent transformation of habitat mosaics (MA 2005, Perrings et al. 2006). The magnitude of impacts of such modifications is expected to be further influenced by other types of environmental shifts e.g. climatic changes, which are forecasted to act synergistically with other land use change promoters (Abildtrup et al. 2006).

Even though some of the most valuable elements of terrestrial biodiversity (e.g. forest core specialists) may be favoured by such changes, it is assumed that agricultural abandonment has a net negative impact on biodiversity because it naturally leads to more homogeneous vegetation cover (Henle et al. 2008). This results from the process of vegetation succession, from an open to a closed landscape, causing: the loss of small-scale mosaics of diversified land use and of their characteristic species, as well as those related to forest edge (ecotone) habitats; a reduction in genetic diversity in both wild species and in local breeds of livestock or varieties of crops; and an increased fire risk in the landscape context, since abandoned grazing areas no longer act as firebreaks (CAP 2004). Agricultural abandonment is also expected to affect the diversity and rates of ecosystem services provided by such homogeneous landscapes (Chytry et al. 2008), while impacts on ecosystem/landscape properties such as resistance to alien species invasion (which is forecasted to be promoted in case of climatic warming; Chytry et al. 2008) are still poorly evaluated.

Agricultural land abandonment also leads to the degradation or disappearance of traditional agricultural infrastructures e.g. terraces (“socalcos”) and traditional irrigation systems (“regadios”). These traditional infrastructures are important for soil preservation and water regulation, and if ignored they will collapse and the system will lose its functionality (Cerqueira et al. 2010). Moreover, the loss of such infrastructures leads to the disappearance of valuable cultural landscapes and traditional ecological knowledge (Cerqueira et al. 2010).

Overall, the increased incidence of land abandonment leads to decreases of landscape complexity, and opens the way for secondary succession following the absence of anthropogenic disturbance (Pereira et al. 2004; Bielsa et al. 2005; Romero-Calcerrada and Perry 2004). Although active management regimes can be applied to control secondary succession and thus minimize habitat and biodiversity loss (García 1992; Muller 2002), this involves a balancing act between the ecological benefits and the financial costs. In considering management strategies, we also need to deliberate who is using those landscapes and for what intention. Presently, many rural mountain communities reveal a lack of dependency of local provisioning services (Pereira et al. 2005). While we would expect a negative impact not only on the landscape but on human well-being, what has been established is that due to the improvement to access to goods and services, local residents no longer have the necessity to work the land (Pereira et al. 2005) and in many cases chose to abandon their rural occupation.

While there is a general belief that in the past rural inhabitants exploited natural resources sustainably, it does not mean that those farming systems have created the best benefits for the environment, economy and human-well being (McMorran et al. 2006; Reed et al. 2013)). In fact, today land abandonment is often seen as an opportunity for nature, providing a break for ecosystems, ecosystem services, biodiversity and ultimately human well-being (Gantioler et al. 2010; Brown et al. 2011). It is seen as an opportunity to correct or rectify the degradation of habitats provoked by the intensification of agriculture over the last century. This is a chance to not only expand the supply of ecosystem services but also reduce the cost of local management strategies (Gantioler et al. 2010; McMorran et al. 2006).

But the question still remains: does abandonment actually benefit ecosystem services and biodiversity? Using a back-casting approach and modeling several ecosystem services, we obtained a glimpse of how particular ecosystem services respond in face of land abandonment (see Chapter 4). By capturing these alterations, using real land cover maps, we were able to obtain a fairly good representation of the impacts abandonment may have on ecosystems and their services. It is a blueprint of these changes which ideally can be used as a future predictor of impacts of on-going land abandonment. Our results highlight that, in face of land abandonment, particular ecosystem services may benefit (i.e. sediment retention and carbon storage). Contrary to land abandonment, studies show that land use intensification is found to have negative impacts on water supply, aesthetic services and biodiversity (Vidal-Legaz et al. 2013; Jiang et al. 2013).

Biodiversity associated to agro-ecosystems may decrease under farmland abandonment, but biodiversity levels remained constant at the landscape level with no decreases for all other taxa (see Chapter 4). This highlights the assumption that many species are capable of adapting to various habitats, with minimal negative impacts due to decreases in agricultural spaces (Proença and Pereira 2013). A recent study revealed that biodiversity changes are mainly a result of changes to community composition rather than species richness at the local level (Dornelas et al. 2014). Local extinction of open habitat specialists has been predicted under abandonment scenarios (Lomba et al. 2013), but this may be managed by maintaining some actively farmed land at the regional scale.

Obtaining a visual representation of the provision of various ecosystem services permits the identification of those regions which are considered important in terms of conservation for future sustainability (Chan et al. 2006; Naidoo et al. 2006; Egoh et al. 2008; Raudsepp et al. 2010; Maes et al. 2012). Many of these regions are associated to those where there is a co-existence of several ecosystem services. Forests, for example, are ecosystems which benefit from land abandonment and where we can find synergies between carbon sequestration and sediment retention, playing a fundamental role in climate regulation and controlling natural hazards (Stoate et al. 2011). Land abandonment leads to the gains of various ecosystem services which are themselves fundamental in underpinning other essential processes and services (Vidal-Legaz et al. 2013; Qiu and Turner 2013; Raudsepp-Hearne et al. 2010). The restoration or enhancement of forest cover, for example, contributes to other essential ecosystem functions, such as high infiltration rate, reducing peak flows and controlling natural hazards and river regimes (Körner 2002; Maes et al. 2009; Santos de Lima et al. 2014). The expansion of native species also contributes to a higher resistance and resilience, contributing to a greater stability of landscapes and therefore of forest processes and ecosystem services (Proença et al. 2010). The protection of forests thus is important, especially of old growth forests, supporting higher levels of biodiversity, water and nutrient services in comparison to young forests (Ferraz et al. 2014; Onaindia et al. 2013). The awareness of the importance of old growth forests has resulted in its incorporation in the EU Biodiversity Strategy through the protection of wilderness (European Commission 2011a). This re-enforces the rationale that restoration or expansion of natural habitats promotes the sustainability of ecological processes while contributing to the new biodiversity strategy.

At the European scale, our findings suggest that restoring or expanding ecosystems suppressed by past agricultural activity positively impacts the supply of various ecosystem services (see Chapter 5). While much skepticism is still found around the topic, abandoned land in mountain systems may well provide the backbone of a sustainable opportunity to balance nature conservation with human well-being. It provides the opportunity to correct or rectify the degradation of habitats provoked by agriculture over the last several decades or centuries. According to our results, projected land abandonment in present agricultural fields would increase the potential supply of various ecosystem services, with minimal negative losses impacts. Nonetheless, this has of course brought forward a number of issues and concerns related to the idea of promoting wilderness. In recent years, land use shifts from agriculture to abandoned land throughout Portugal and other European countries resulted in increased fire susceptibility (Moreira et al. 2001). In the past, extensive marginal lands were cleared in order for herds and cattle to graze and pass through, and wood was gathered for human use. Due to the new demographic structure these areas are now covered with dense heath and scrub, thereby increasing fire occurrence (Shakesby 2011), which consequently promotes soil erosion and land degradation (León et al. 2014; Inbar et al. 2014).

6.4 Future perspectives for changing rural landscapes

Even though there is still a large amount of uncertainty concerning the responses of ecosystems and their services to ongoing environmental changes, particularly concerning climatic shifts, land use change, and their synergetic effects with other drivers (Schröter et al. 2005; Metzger et al. 2006), conservation perspectives are fairly positive throughout Europe, where ecosystems and ecosystem services have been identified as the core focus of land management and conservation planning, including the new biodiversity strategy (European Commission 2011a).

Under target 2 of the new biodiversity strategy, incentives have been created to encourage investment in green infrastructure and maintenance of ecosystem services. This strategy not only focuses on restoring ecosystems and their services, but its targets are expanded to include the contribution of agriculture in enhancing biodiversity and services, through the direct payments for environmental goods in the EU Common Agricultural Policy (European Commission 2011b). It also offers a possibility for those residents who want to remain in mountain areas to explore a different type of income through payments for ecosystem services (PES). Nonetheless, we are still faced with the dilemma of those regions which have undergone severe rural exodus and those which have continued to suffer with out migration, and where the implementation of incentives for management plans may not be as affective or realistic. These regions are mainly mountain systems where there have been changes to human demand for particular services (Harrison et al. 2010). During this time of abandonment, the quality of some ecosystem services has improved, primarily those associated to forest ecosystems (i.e. timber production, freshwater provision, erosion and natural hazard regulation, and recreation) (Harrison et al. 2010).

Mountain systems and other high elevation regions are also linked to wilderness areas (Carver 2010). Due to the low representation of wilderness areas across Europe, efforts are now focused on the expansion of wild areas (i.e. PAN Parks). It´s recognized that wild areas provide a wide range of ecosystem services and are self-sustaining (Costanza and Mageau 1999). Undisturbed natural habitats are known to supply higher levels of services (Schils et al. 2008). Our findings highlight that high elevation regions supply higher levels of ecosystem services and primarily regulating and cultural benefits (see Chapter 5). Contrary, a high supply of provisioning services is mainly associated to low wilderness.

Although studies have shown the benefits of wilderness areas (e.g. McMorran et al. 2006; Shils et al. 2008; Brown et al. 2011), we were still lacking assessments of the potential gains and losses of ecosystem services through the promotion of rewilding. While our attempt to determine the impacts on ecosystem services is preliminary, we were able to assess the impact of land abandonment or rewilding in agricultural areas. In increasing the size and connectivity of high quality wilderness we were able to obtain positive trends for various indicators. Although a recent concept throughout Europe, rewilding is seen as an opportunity where historical social-ecological systems are no longer socially, economically or ecologically beneficial.

However, when considering the restoration of a particular area after farming abandonment, several factors need to be considered. From a holistic perspective, both passive and active land restoration in Europe needs to be viewed as a management strategy that provide benefits to humans, through the flow of ecosystem services, while preserving biodiversity and potentially generating economic relief to local and regional populations (McMorran et al. 2006; Brown et al. 2011). Nonetheless, the application of these two management strategies is dependent on the state of the ecosystem (Chazdon 2008), the climatic, biophysical, socio-economic elements of the location, as well as the costs and benefits associated to each management option.

In fact, both active and passive management strategies present advantages and disadvantages which should be considered and evaluated for each specific case. Active restoration enables a faster recuperation of natural functions of the ecosystem and consequently decreases the time lag of the economic benefits obtained from ecosystem services (Chazdon 2008). However, it implies a higher cost of management. Conversely, passive management has a slower recovery period and low management cost. Both are efficient in terms of biodiversity and provision of ecosystem services (Rey Benayas et al. 2009). Nonetheless, we cannot discard the on-going paradox of how much human participation is accepted in the restoration of wilderness and that much of Europe´s rural landscapes are linked to social-ecological systems which have evolved through time and have generated cultural roots and historical value (Walker and Ryan 2008).

The degradation of natural ecosystems leads to exacerbated consequences in the provision of ecosystem services, impacting our economy and essential human well-being. The increased vulnerability of ecosystem services to land use change in Europe has led to policy targets whose initiatives aim at the preservation and restoration of natural vegetation that promotes the conservation of biodiversity naturally underpinning the flow of ecosystem services. The PAN Parks initiative has recently highlighted several financial benefits associated to Europe´s wilderness and how payment for ecosystem services could potentially be integrated in protected areas (Houdet 2011).

Today, maintaining ecosystem services contributing to human well-being while conserving biodiversity and other natural services in agricultural landscapes has become a global priority (European Commission 2011a; Benayas et al. 2009; Dunbar et al. 2013). Although agricultural activity is considered a major cause of destruction or degradation of natural ecosystems, we are dependent on these agro-ecosystems for food production and other services. A challenge which has been much discussed is the reconciliation of food production and biodiversity through two different approaches – land sparing and land sharing (Phalan et al. 2011). These two types of interventions represent different objectives for food production and biodiversity. In land sharing, both biodiversity conservation and food production are retained in the same land through methods that are nature friendly. Conversely, land sparing consists of disconnecting land targeted for conservation from land for agricultural production. According to Navarro and Pereira (2012), both concepts are necessary in future rewilding efforts. However, much skepticism has been built around the topic and many believe that land sparing will fail to promote the conservation of wild biodiversity in the face of climate change by reducing the connectivity between habitats fundamental in the protection of various species. These concepts need to be further explored and essential questions need to be answered. At what scale do we apply and integrate the two concepts? Should these approaches be considered simultaneously? In Rey Benayas and Bullock (2012), both concepts are explored and a suggested solution to some of the challenges is the creation of woodland islets, combining the two interventions to maximize food provision, biodiversity and conservation.

6.5. Challenges ahead and future research

Agricultural land abandonment is a problem that can no longer be ignored and one which will affect many mountainous regions and other marginal areas. It is expected to increase throughout Europe (Keenlyside and Tucker; Navarro and Pereira 2012), and based on our findings we argue that the ecological restoration (both active and passive) of these regions is an opportunity for biodiversity and ecosystem services (Navrro and Pereira 2012; ReyBenayas and Bullock 2012; see Chapter 4 and 5). By protecting and increasing wilderness we are underpinning areas of high ecosystem service provision. However, we cannot ignore the drawbacks and trade-offs associated to rewilding. Changes in agriculture may cause a local loss of species-rich ecosystems that depend on traditional land use (Poschlod et al. 2005). Particularly on High Nature Value (HNV) farmland, abandonment may involve significant losses of biodiversity, because many of its characteristic species strongly depend on human management with low inputs of fertilizers as well as on grazing or mowing (CAP 2004; Lomba et al. 2013).

Different management regimes can be applied to control secondary succession and thus minimize habitat and biodiversity loss (García 1992, Muller 2002). In practice, managers have to compromise between ecological benefits and financial costs of management schemes. Furthermore, we have yet to determine how this would affect social-ecological systems, primarily human's adaptive capacity to changes in the local provision of ecosystem services. This emerging balance calls for further research and increase awareness of the potential social, environmental and economic values associated to wilderness areas. Determining how local populations could benefit may help promote the concept and reinforce the idea of naturalness as an opportunity to promote human well-being and to support public policies and funding of nature conservation (McMorran et al. 2006; Brown et al. 2011; Hein et al. 2011).

The research developed for this thesis has provided relevant results and conclusions, but it has also highlighted pertinent questions for future investigation. In light of these findings, it is evident that there is a lack of information when it comes to selecting the most appropriate management plan in face of land abandonment. Further research is also required to determine local inhabitants' perception on rewilding efforts as well as other management strategies. Realistically, all of Europe cannot be wilderness, therefore future research should focus on determining "rewilding hotspots". Determining the benefits and costs of the various strategies are imperative and should be locally/regionally specific. Furthermore, determining local inhabitant's involvement in managing the land, its drivers and frictions, and establishing incentives that not only promotes the provision of multiple ecosystem services but also contribute to global biodiversity loss, are issues that need to be addressed more thoroughly. Additionally, determining local inhabitants' sensitivity to various management strategies is essential, as is promoting educational programs outlining the various benefits of restoration.

In general, selecting a strategy that creates a positive interaction between society and the environment is fundamental in promoting multi-functional landscapes, mainly in rural mountain areas (Carvalho-Ribeiro and Lovett 2010;Surová and Pinto-Correia 2008). Mapping ecosystem services at different scales is essential for any knowledge-driven management strategy. Although we are seeing an increasing number of studies focusing on the mapping of ecosystem services, we are still lacking fundamental data on ecosystem services as well as a consensus of the various indicators for a number of services. An example are the limited indicators for cultural services (Maes et al. 2011).

Framing policies that target the safeguarding of the systems ecology, while promoting social and economic benefits is of an essence, mainly in mountain areas, as human well-being over wide regions depends on the sustainability of ecosystem services provided by these landscapes. It is therefore fundamental we consider the ecological impacts that the post-abandonment succession, natural hazards and the loss of species with high affinity to agro-ecosystems will have locally, regionally and globally. Finally, a strong emphasis should be put on assessing the effects of changes in landscape heterogeneity on the loss of ecosystem services, resilience and adaptive capacity of social-ecological systems at the several relevant scales (Chapin III et al. 2009).

6.6. References

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Abildtrup, J., Audsley, E., Fekete-Farkas, M., Giupponi, C., Gylling, M., Rosato, P. & Rounsevell, M. (2006). Socio-economic scenario development for the assessment of climate change impacts on agricultural land use: a pairwise comparison approach. *Environmental Science & Policy* 9: 101-115.

Bassi, M., & Tache, B. (2011). The community conserved landscaped of the BoranaOromo, Ethiopia. *Management of Environmental Quality, 22*(2), 174-186.

Beilin, R., Lindborg, R., Stenseke, M., Pereira, H. M., Llausàs, A., Slätmo, E., Cerqueira, Y., Navarro, L., Rodrigues, P., Reichelt, N., Munro, N. & Queiroz, C. (2014). Analysing how drivers of agricultural land abandonment affect biodiversity and cultural landscapes using case studies from Scandinavia, Iberia and Oceania. Land Use Policy, 36:60-72.

Benayas, J. M. R., Newton, A. C., Diaz, A., & Bullock, J. M. (2009). Enhancement of Biodiversity and Ecosystem Services by Ecological Restoration: A Meta-Analysis. *Science*, *325*(5944), 1121–1124.

Bielsa, I., Pons, X. and Bunce, B. (2005). Agricultural Abandonment in the North Eastern Iberian Peninsula: The Use of Basic Landscape Metrics to Support Planning. Journal of Environmental Planning and Management 48(1), pp. 85–102 doi:10.1080/0964056042000308166

Bignal, E.M. & McCracken, D.I. (1996). Low-intensity farming systems in the conservation of the countryside. *Journal of Applied Ecology* 33: 413–424.

Brown, C., Mcmorran, R., & Price, M. F. (2011). Rewilding – A New Paradigm for Nature Conservation in Scotland? *Scottish Geographical Journal*, *127*(4), 288–314. doi:10.1080/14702541.2012.666261

Bürgi, M., Hersperger, A., & Schneeberger, N. (2004). Driving forces of landscape change — current and new directions. *Landscape Ecology*, *19*(8), 857–868. doi:10.1007/s10980-004-0245-8

CAP, 2004. *Common Agricultural Policy - Seminar on Land Abandonment and Biodiversity, in relation to the 1st and 2nd pillars of the EU's Common Agricultural Policy*. Sigulda, Latvia, 7-8 October 2004.

Carvalho, A.M. and Frazão-Moreira, A. (2011). Importance of local knowledge in plant resources management and conservation in two protected areas fromTrás-os-ontes,Portugal. *Journal of Ethnobiology and Ethnomedicine* 7: 36.

Carvalho-Ribeiro, S. M., Lovett, A., O´Riordan, T. (2010). Multifunctional forest management in Northern Portugal: Moving from scenarios to governance for sustainable development. Land Use Policy, 27(4):1111-1122.

Cerqueira, Y. C., Araújo, C., Vincente, J., Pereira, H. M. and Honrado, J., 2010. Ecological and cultural consequences of agricultural abandonment in the Peneda-Gerês National Park (Portugal). In: T. Evelpidou, T. Figueiredo, F. Mauro, V. Tecim, A. Vassilopoulos. (Eds), *Natural Heritage from East to West: Case studies from 6 EU countries* (pp. 175-183). London, New York.

Carver, S. (2010). Chapter 10.3 Mountains and wilderness, in European Environment Agency Europe´s ecological backbone: Recognizing the true value of our mountains, EEA Report No 6/2010, pp. 192-201.

Chan, K. M. A., Shaw, M. R., Cameron, D. R., Underwood, E. C., & Daily, G. C. (2006).

Conservation Planning for Ecosystem Services. *PLoS Biol*, *4*(11), e379. doi:10.1371/journal.pbio.0040379

Chapin III, F.S., Carpenter, S.R., Kofinas, G.P., Folke, C., Abel, N., Clark, W.C., Olsson, P., (...), Swanson, F.J. (2010) Ecosystem stewardship: sustainability strategies for a rapidly changing planet Trends in Ecology and Evolution, 25 (4) , pp. 241-249.

Chazdon, R. L. (2008). Beyond deforestation: Restoring forests and ecosystem services on degraded lands. Science, 320:1458-1460.

Chytry, M., Maskell, L.C., Pino, J., Pysek, P., Vilà, M., Font, X. & Smart, S.M. (2008). Habitat invasions by alien plants: a quantitative comparison among Mediterranean, subcontinental and oceanic regions of Europe. *Journal of Applied Ecology* 45: 448-458.

Dornelas, M., Gotelli, N. J., McGill, B., Shimadzy, H., Moyes, F., Sievers, C., Magurran, A. (2014). Assemblage Time Series Reveal Biodiversity Change but Not Systemic Loss. Science 344, 296.

Dunbar, M. B., Panagos, P., & Montanarella, L. (2013). European perspective of ecosystem services and related policies. *Integrated Environmental Assessment and Management*, *9*(2), 231–236. doi:10.1002/ieam.1400

Egoh, B., Reyers, B., Rouget, M., Richardson, D. M., Le Maitre, D. C., & van Jaarsveld, A. S. (2008). Mapping ecosystem services for planning and management. *Agriculture, Ecosystems & Environment*, *127*(1–2), 135–140. doi:10.1016/j.agee.2008.03.013

Emery, Marla R.; Wrobel, Alexandra; Hansen, Mark H.; Dockry, Michael; Moser, W. Keith; Stark, Kekek Jason; Gilbert, Jonathan H. (2014). Using traditional ecological knowledge as a basis for targeted forest inventory: paper birch (*Betula papyrifera*) in the US Great Lakes Region. Journal of Forestry 112(2): 207-214.

European Commission (2011a). Our life insurance, our natural capital: an EU biodiversity strategy to 2020. Brussels, 2011-05-03. COM(2011) 244.

European Commission (2011b). Our life insurance, our natural capital: an EU biodiversity strategy to 2020. Brussels, 2011-05-03. COM(2011) 244.

Ferraz, S. B., Ferraz, K. P. M. B., Cassiano, C., Brancalion, P., Luz, D. A., Azevedo, T., … Metzger, J. (2014). How good are tropical forest patches for ecosystem services provisioning? *Landscape Ecology*, *29*(2), 187–200. doi:10.1007/s10980-014-9988-z

Figueiredo J, Pereira HM (2011) Regime shifts in socio-ecological model of farmland abandonment. Landscape Ecol 26: 737–749 doi:10.1007/s10980-011-9605-3.

Gantioler, S., Rayment, M., Bassi, S., Kettunen, M., McConville, A., Landgrebe, R., Gerdes, H. & ten Brink, P. (2010) *Costs and Socio-Economic Benefits associated* *with the Natura 2000 Network.* Final report to the European Commission, DG Environment on Contract ENV.B.2/SER/2008/0038. Institute for European Environmental Policy / GHK / Ecologic,Brussels 2010

García, A. (1992). Conserving the species-rich meadows of Europe. *Biotic Diversity in Agroecosystems*, *40*(1–4), 219–232. doi:10.1016/0167-8809(92)90094-R

Glasenapp, M., & Thornton, T. (2011). Traditional Ecological Knowledge of Swiss Alpine Farmers and their Resilience to Socioecological Change. *Human Ecology*, *39*(6), 769–781. doi:10.1007/s10745-011-9427-6

Gobster PH, Nassauer JI, Daniel TC, Fry G. (2007). The shared landscape: what does aesthetics have to do with ecology? Landsc Ecol 22:959–72

Gómez-Baggethun, E., S. Mingorria, V. Reyes-García, L. Calvet and C. Montes. (2010). Traditional ecological knowledge trends in the transition to a market economy: Empirical study in the Doñana natural areas. *Conservation Biology* 24: 721-729.

Gutman, P. (2007). Ecosystem services: Foundations for a new rural–urban compact. *Ecological Economics*, *62*(3–4), 383–387. doi:10.1016/j.ecolecon.2007.02.027

Halada, L., Evans, D., Romão, C., & Petersen, J.-E. (2011). Which habitats of European importance depend on agricultural practices? *Biodiversity and Conservation*, *20*(11), 2365–2378. doi:10.1007/s10531-011-9989-z

Harrison, P., Vandewalle, M., Sykes, M., Berry, P., Bugter, R., Bello, F., … Zobel, M. (2010). Identifying and prioritising services in European terrestrial and freshwater ecosystems. *Biodiversity and Conservation*, *19*(10), 2791–2821. doi:10.1007/s10531-010-9789-x

Henle, K., Alard, D., Clitherow, J., Cobb, P., Firbank, L., Kull, T., McCrackeng, D., Moritz, R.F.A., Niemelä, J., Rebane, M., Wascher, D., Watt, A. & Young, J. (2008). Identifying and managing the conflicts between agriculture and biodiversity conservation in Europe - A review. *Agriculture, Ecosystems and Environment* 124: 60-71.

Hunziker, M., Felber, P., Gehring, K., Buchecker, M., Bauer, N., & Kienast, F. (2008). Evaluation of Landscape Change by Different Social Groups. *Mountain Research and Development*, *28*(2), 140–147. doi:10.1659/mrd.0952

Inbar, A., Lado, M., Sternberg, M., Tenau, H., Ben-Hur, M. (2014). Forest fire effects on soil chemical and physiochemical properties, infiltration, runoff, and erosion in a semiarid Mediterranean region. Geodema, 221:131-138.

Jiang, M., Bullock, J. M., & Hooftman, D. A. P. (2013a). Mapping ecosystem service and biodiversity changes over 70 years in a rural English county. *Journal of Applied*

Ecology, *50*(4), 841–850. doi:10.1111/1365-2664.12093

Kaltenborn BP, Bjerke T. (2002). Association between environmental value orientations and landscape preferences. Landscape and Urban Planning 59:1–11.

Keenleyside C. & Tucker, G. (2010). Farmland Abandonment in the EU: an assessment of rends and prospects. London: WWF and IEEP. 97 pp.

Kimmerer, R. W. (2012). Searching for synergy: integrating traditional and scientific ecological knowledge in environmental science education. Journal of Environmental Studies and Sciences, 2(4):317-323.

Körner, C. and Ohsawa, M. (eds.), 2005. Mountain systems. *In:* Hassan, R.; Scholes, R. and Ash, N. (eds.), *Ecosystems and Human Well-being: Current State and Trends, Volume 1.* Island Press, Washington DC: 681–716.

León, J., Cerdà, A., Badía, D., Echeverría, M., Martí, C. 2014. Analyzing the vegetation response under different treatments after wildfires in NE Spain. Geophysical Research Abstracts.

Lindenmayer, D.B. & Franklin, J.F. 2002. *Conserving forest biodiversity: a comprehensive multiscaled approach.* Island Press, Washington, D.C.

Lomba, A., Gonçalves, J., Moreira, F., Honrado, J. (2013).Simulating long-term effects of abandonment on plant diversity in mediterranean mountain farmland. Plant Biosystems 147: 328-342.

MA (Millennium Ecosystem Assessment) 2005. Ecosystem and human well-being. Synthesis. Island Press. Washington D. C. 137 p.

Maes, J., W. H., Heuvelmans, G., Muys, B. (2009). Assessment of land use impact on water related ecosystem service capturing the integrated terrestrial aquatic-system. Environmental Science of Technology, 43:7324-7330.

Maes, J., Paracchini, M.L., & Zulian, G. (2011). A European Assessment of the Provision of Ecosystem Services: Towards an Atlas of Ecosystem Services. Publications Office of the European Union, Luxembourg, doi:10.2788/63557, p. 81.

Maes, J., Hauck, J., Paracchini, M. L., Ratamäki, O., Termanse, M., Perez-Sboa, M., Kopperoinen, L., Rankinen, K., Schägner, J. P., Henrys, P., Cisowska, I., Zandersen, M., Jax, K., La Notte, A., Leikola, N., Pouta, E., Smart, S., Hasler, B., Lankia, T., Andersen, H. E., Lavalle, C., Vermaas, T., Alemu, M. H., Scholefield,P., Batista, F., Pywell, R., Hutchins, M., Blemmer, M., Fonnesbech-Wulff, A., Vanbergen, A. J., Münier, B., Baranzelli, C., Roy, D., Thieu, V., Zulian, G., Kuussaari, M., Thodsen, H., Alanen, E.L., Egoh, B., Sørensen, P. B., Braat, L. & Bidoglio, G. (2012b). *A spatial assessment of ecosystem services in Europe:*

methods, case studies and policy analysis. - phase 2. PEER Report No 4. Ispra: Partnership for European EnvironmentalResearch

McMorran, R., Price, M. F. & McVittie, A. (2006). A review of the benefits and opportunities attributed to Scotland´s landscapes of wild character. Scottish Natural Heritage Commissioned Report No. 194 (ROAME No. F04NC18).

Metzger, M. J., Rounsevell, M. D. A., Acosta-Michlik, L., Leemans, R., & Schröter, D. (2006). The vulnerability of ecosystem services to land use change. *Scenario-Based Studies of Future Land Use in Europe*, *114*(1), 69–85. doi:10.1016/j.agee.2005.11.025

Moreira, F., Rego, F., & Ferreira, P. (2001). Temporal (1958–1995) pattern of change in a cultural landscape of northwestern Portugal: implications for fire occurrence. *Landscape Ecology*, *16*(6), 557–567. doi:10.1023/A:1013130528470

Muller, S. (2002). Diversity of management practices required to ensure conservation of rare and locally threatened plant species in grasslands: a case study at a regional scale (Lorraine, France). *Biodiversity & Conservation*, *11*(7), 1173–1184. doi:10.1023/A:1016049605021

Naidoo, R., & Ricketts, T. H. (2006). Mapping the Economic Costs and Benefits of Conservation. *PLoS Biol*, *4*(11), e360. doi:10.1371/journal.pbio.0040360

Natori, Y., Chenoweth, R. (2008). Differences in rural landscape perceptions and preferences between farmers and naturalists. Journal of Environmental Psychology 28, 250-267.

Nijnik, M., Zahvoyska, L., Nijnik, A., & Ode, A. (2009). Public evaluation of landscape content and change: Several examples from Europe. *Formalisation of Land Rights in the South*, *26*(1), 77–86. doi:10.1016/j.landusepol.2008.03.001

Onaindia, M., Fernández, D. M., Madariaga, I., Rodríguez-Loinaz, G. (2013). Co-benefits and trade-offs between biodiversity, the carbon storage, and water flow regulations. Forest Ecology Management, 289:1-9.

Pereira, H.M., Domingos, T. & Vicente, L. eds. 2004. *Portugal Millennium Ecosystem Assessment: State of the assessment report.* Centro de Biologia Ambiental, Faculdade de Ciências da Universidade de Lisboa.

Pereira, E., Queiroz, C., Pereira, H. M., & Vicente, L. (2005). Ecosystem Services and Human Well-Being: a Participatory Study in a Mountain Community in Portugal. *Ecology and Society*, *10*(2). Retrieved from http://www.ecologyandsociety.org/vol10/iss2/art14/

Perrings, C., Jackson, L., Bawa, K., Brussaard, L., Brush, S., Gavin, T., Papa, R., Pascual, U. & De Ruiter, P. (2006). Biodiversity in agricultural landscapes: saving natural capital without losing interes. Conservation Biology 20(2): 263-264.

Phalan, B., Onial, M., Balmford, A., Green, R. E. (2011). Reconciling food production and biodiversity conservation:land sharing and land sparing compared. Science, 33, 1289.

Plieninger, T., and C. Bieling. (2013). Resilience-based perspectives to guiding high-nature-value farmland through socioeconomic change. *Ecology and Society* **18**(4): 20. http://dx.doi.org/10.5751/ES-05877-180420

Poschlod, P., Bakker, J. P., & Kahmen, S. (2005). Changing land use and its impact on biodiversity. *Basic and Applied Ecology*, *6*(2), 93–98. doi:10.1016/j.baae.2004.12.001

Proença, V., & Pereira, H. M. (2013). Species–area models to assess biodiversity change in multi-habitat landscapes: The importance of species habitat affinity. *Basic and Applied Ecology*, *14*(2), 102–114. doi:10.1016/j.baae.2012.10.010

Qiu, J., & Turner, M. G. (2013). Spatial interactions among ecosystem services in an urbanizing agricultural watershed. *Proceedings of the National Academy of Sciences*. Retrieved from http://www.pnas.org/content/early/2013/06/27/1310539110.abstract

Raudsepp-Hearne, C., Peterson, G. D., Bennett, E. M. (2010). Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. PNAS, 1-6.

Reed, M. S., Hubacek, K., Bonn, A., Burt, T. P., Holden, J., Stringer, L. C., … Worrall, F. (2013). Anticipating and Managing Future Trade-offs and Complementarities between Ecosystem Services. *Ecology and Society*, *18*(1). doi:10.5751/ES-04924-180105

Reidsma, P., Tekelenburg, T., van den Berg, M., & Alkemade, R. (2006). Impacts of land-use change on biodiversity: An assessment of agricultural biodiversity in the European Union. *Scenario-Based Studies of Future Land Use in Europe*, *114*(1), 86–102. doi:10.1016/j.agee.2005.11.026

Rey Benayas, J. M., Martins, A., Nicolau, J. M. and Shulz, J. J. 2007. Abandonment of agricultural land: an overview of drivers and consequences. CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources, 2(57):1-14.

Rey Benayas J, Bullock J. (2012). Restoration of biodiversity and ecosystem services on agricultural land. Ecosystems. doi:10.1007/s10021-012-9552-0.

Romero-Calcerrada, R., & Perry, G. L. W. (2004). , Central Spain, 1984–1999. *Landscape and Urban Planning*, *66*(4), 217–232. doi:10.1016/S0169-2046(03)00112-9

Santos de Lima L., Coe, M.T., Soares-Filho, B.S., Cuadra, S.V., Pinto Dias, L.C., Costa, M.H,. Santos de Lima, L., Rodrigues, H.O .(2014). Feedbacks between deforestation, climate, and hydrology in the Southwestern Amazon: implications for the provision of ecosystem services. Landscape Ecol. doi:10.1007/s10980-013-9962-1

Sayadi, S., González-Roa, M. C., & Calatrava-Requena, J. (2009). Public preferences for landscape features: The case of agricultural landscape in mountainous Mediterranean areas. *Land Use Policy*, *26*(2), 334–344. doi:10.1016/j.landusepol.2008.04.003

Scherr, S.J. & McNeely, J.A. (2008). Biodiversity conservation and agricultural sustainability towards a new paradigm of 'ecoagriculture' landscapes. *Phil.Trans. R. Soc. B* 363: 477-494.

Schröter, D., Cramer, W., Leemans, R., Prentice, I. C., Araujo, M. B., Arnell, N. W., Bondeau, A., Bugmann, H., Carter, T. R., Gracia, C. A. (2005). Ecosystem service supply and vulnerability to global change in Europe. Science, 310:1333–7.

Shakesby, R. A. (2011). Post-wildfire soil erosion in the Mediterranean: Review and future research directions. Earth Science Reviews, 105:71-100.

Sirami, C., Brotons, L., & Martin, J.-L. (2007). Vegetation and songbird response to land abandonment: from landscape to census plot. *Diversity and Distributions*, *13*(1), 42–52. doi:10.1111/j.1472-4642.2006.00297.x

Soliva, R., Hunziker, M. (2009). How do biodiversity and conservation values relate to landscape preferences? A case study from the Swiss Alps. Biodivers. Conserv.18, 2483-2507.

Surová, D. and Pinto-correia, T. (2008). Landscape preferences in the cork oak Montado region of Alentejo, southern Portugal: Searching for valuable landscape characteristics for different user groups', Landscape Research, 33:3, 311 — 330

Van den Berg AE, Koole SL. (2006). New wilderness in the Netherlands: An investigation of visual preferences for nature development landscapes. Landscape and Urban Planning 78:362–372.

Verburg, P., & Overmars, K. (2009). Combining top-down and bottom-up dynamics in land use modeling: exploring the future of abandoned farmlands in Europe with the

Dyna-CLUE model. *Landscape Ecology*, *24*(9), 1167–1181. doi:10.1007/s10980-009-9355-7

Vidal-Legaz, B., Martínez-Fernández, J., Sánchez Picón, A., Pugnaire, F. I. (2013). Trade-offs between maintenance of ecosystem services and socio-economic development in rural mountainous communities in Southern Spain: A dynamic simulation approach. Journal of Environmental Management 131 (2013) 280e297

Walker, A. J. and Ryan, R. L. (2008). Place attachment and landscape preservation in rural New England: A Maine case study. Landscape and Urban Planning 86(2), pp. 141–152.